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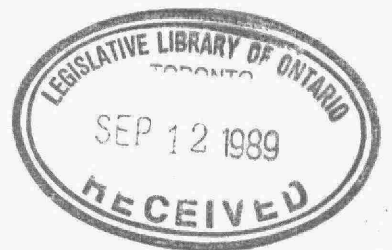
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ANALYSIS OF THE CONTAMINANT PLUME

IN THE OAK RIDGES AQUIFER

R. A. C. PROJECT NO. 261 RR

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Abstract

The Stouffville landfill site is situated on the Oak Ridges Moraine, a major topographical feature marking the drainage divide between Lake Ontario and Lake Simcoe. Three main hydrogeological units have been identified at the site: the Halton Till at the surface; the Oak Ridges Aquifer; and a lower aquitard comprised of three different fine-grained glacial deposits. The latter effectively marks the base of the flow system of relevance in this study.

Flow in the main aquifer is strongly influenced by recharge at Musselman Lake and associated swamps located along the north side of the landfill. Groundwater flow is generally to the south and southwest from these major recharge zones. Capping the main aquifer unit is the Halton Till which is the receiving unit for all the waste, both solid and liquid, disposed of at the landfill. Permeable pathways through the till have allowed contaminants to reach the Oak Ridges Aquifer and develop a contaminant plume. No significant off-site migration of contaminants, however, occurs within the Halton Till.

Contaminant peaks measured in samples from observation wells in the Oak Ridges Aquifer downgradient of the site show trends which have been attributed to the early practice of dumping industrial liquid waste into "lagoons" occupying small kettle depressions which drained very rapidly. Tritium results indicate the presence of pre-1953 recharge at depth in the aquifer, and locally in shallower zones. The concentrations inside the chloride plume can be explained if the origin of the plume is liquid waste that is free of bomb tritium.

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Variation in Cl⁻/species ratios approximately along the center of the plume indicate that cation exchange and sulphate attenuation occur in the plume.

The most appropriate inorganic constituent to delineate the plume in the main aquifer is chloride as it shows the best contrast with background water quality. The contaminant plume extends 750 m off the southwest corner of the site and, in plan view, it is approximately elliptic and contains several zones of relatively high contaminated groundwater downgradient from different lagoons. From 1984 to 1986, the plume migrated about 70 m farther downgradient, its size increased slightly and the peak values decreased.

The estimated average migration velocity of 16 to 38 m/yr, obtained by curve fitting of breakthrough curves with the analytical transport model, is compatible with the migration of the plume contoured at different times. The longitudinal dispersivity estimation of 15 to 40 is in the range mentioned in the literature for similar aquifers and plumes.

The level of different parameters was contoured in two aquifer cross sections. In the first one, the plume delineated was orientated downward and divided into two zones downgradient. In the second cross section, two horizontal plumes were contoured, suggesting that heterogeneities in the aquifer caused a separation of the plume.

The application of a flow model in a cross section confirms the large influence of heterogeneities in the aquifer on the flow patterns, as expected from the plume configuration. Simulations of recharge conditions during the active years of the landfill shows a rise in the water table of a few metres,

causing an increase in the horizontal flow velocity. The plume resulting from advection only in the simulated flow field, assuming a higher concentration of contaminants in the recharge from the lagoons than from the landfill, has two inside cores as delineated with several parameters. If current conditions prevail, the model suggests that, in the future, contamination will be limited to a shallow portion of the aquifer.

Results from the transport model lead to the prediction that the peak values in the chloride plume caused by liquid waste disposal will continue to decrease. However, the contamination from solid waste has yet to impact the aquifer. The plumes resulting from leachate infiltration should migrate downgradient at a velocity lower than 40 m/yr.

Introduction

Waste Management Incorporated Site #4 (to be referred to as Site 4), also referred to as the Stouffville landfill, is located about 40 km northeast of Toronto. Site 4 was closed in July 1985, after receiving more than 10 million tons of solid and liquid industrial waste over the course of more than 20 years. Fear of contamination of groundwater supplies from the Oak Ridges Aquifer caused considerable controversy over the hazard presented by the disposal operations. Rather extensive field studies, complemented by sophisticated laboratory studies, were made to provide evidence that Site 4 could continue operations without causing undue harm and then (in 1983), to provide the information required to close the site according to an agreement between the Ontario Ministry of the Environment (MOE); the owner Waste Management Incorporated (WMI); and the Town of Stouffville.

The Oak Ridges Aquifer provides water of excellent quality to municipal, industrial and domestic wells throughout the region. Musselman Lake and associated swamps have been identified as important recharge zones (Kaye, 1986; Kaye and Farvolden, 1988). The discharge zone is along the southern flank of the moraine where flowing wells are common and springs are the source of water for numerous ponds and streams. Local contamination of this valuable resource stems from solid wastes buried in the Halton Till and thousands of cubic metres of liquid industrial wastes dumped in kettles at the landfill site, called "lagoons".

Beginning in 1966, a series of reports on different aspects of the hydrogeology at Site 4 were prepared for the owner, first by Hydrology Consultants Limited (HCL), and then by Conestoga Rovers and Associates (CRA). These early reports, and reports from the Environmental Assesment Board public hearings of 1975 and 1981 on Site 4, were reviewed by G.M. Hughes from the Ministry of the Environment. His report (MOE, 1982a) provided an excellent summary of the situation up to 1980.

In November 1981, bacterial mutagenicity in the water of a private well adjacent to Site 4 was reported by a researcher at the University of Western Ontario. This brought about the initiation by the MOE in December 1981 of a state-of-the-art chemical and biological study of groundwater at the site The final report showed no evidence of bacterial mutagenigenic activity (MOE, 1982b).

From 1982 to 1984, a hydrogeologic investigation was conducted by CRA for the owner as a condition for completion of activities and closing of the landfill site (CRA, 1985). In order to extend the hydrogeological interpretations by CRA, Kaye (1986) studied the role of nearby Musselman Lake in the hydrogeology of the area northeast of the site.

Since 1970, a total of 107 observation wells and leachate wells have been installed at the site. Of these installations, 79 were completed in the main aquifer and 17 boreholes penetrated the entire aquifer thickness. Drilling methods employed include cable tool, mud rotary, and hollow stem auger.

Geophysical logs were obtained from the 17 "deep" drill holes and used to determine the hydrostratigraphy of the site. Electrical logs were used to

An interesting characteristic of this site is its hydrogeological setting, a water table aquifer apparently partially confined by a till of low permeability. The limited amount of recharge under such conditions has important implications with respect to contamination dilution. Studies of plumes in aquifers influenced by recharge at the water table are available in the literature. Information is however more sparse on contaminant migration in the conditions observed in the Oak Ridges Aquifer.

In this study, the results of the monitoring program are used to follow the extent of the zone of contaminated water with time and to delineate its shape. The hydrochemical processes occurring in the plume are studied. Tritium (^3H) is used to delineate zones of groundwater of different ages and relate these zones to plume development. The influence of recharge and heterogeneous stratigraphy in the aquifer on the plume is tested with flow model simulations. An analytical transport model is used to obtain estimates of the hydrologic factors controlling the plume evolution.

Previous studies

Surface geology of the Oak Ridges Moraine was dicussed by Chapman and Putnam (1966) and mapped in detail by Gwyn and White (1973); Gwyn and DiLabio (1973); and White (1975). The depositional history of the western portion of the moraine was described by Duckworth (1979). Groundwater resources in the Duffins Creek-Rouge River drainage basins where springs from the Oak Ridges Aquifer form the headwaters of streams, downgradient of Site 4, were described by Sibul et al (1982).

obtain apparent resistivities and stratigraphic contacts. Natural gamma logs were used to estimate clay content and caliper logs measured the hole diameters. Split spoon samples of selected zones of two of the deep holes were collected to improve interpretation of the logs.

Numerous grain size analyses were performed on split spoon and test pit samples from the Halton Till (56); the main aquifer (40); and the three under-lying units (63).

Most of the groundwater monitoring devices on site consisted of 1.5 m long (approximately) stainless steel screen attached to the bottom of a 3.8 to 5.1 cm ($1^1/_2$-$2^1/_2$") diameter black iron pipe. A sandpack was naturally developed where possible, otherwise commercial silica sand was installed over an approximately 6 m long interval.

Early piezometer nests consisted of 2 or 3 standpipes installed in individual boreholes a few metres apart. Since 1983, 2 or 3 piezometer pipes have been installed in the same borehole with the different piezometer intakes distributed through the aquifer thickness. The first was placed close to the aquifer bottom; the second in the central part and the third about 7 m below the water table. Screens were isolated by installation of bentonite or cement/ bentonite grout above and below the sandpack.

Purpose and scope of the report

The specific objectives of this study are: (1) to describe the hydrogeological environment of the landfill site; (2) to describe the evolution of the contaminant plume; (3) to analyze the plume behavior with a) physical flow and

transport models and b) hydrogeochemical and isotopic studies; and (4) to develop a predictive model of future plume behaviour.

Previous studies of the closed WMI Site 4 were reviewed to provide a description of the operations and the physical and hydrogeological environments at the site. Borehole logs, geophysical logs, water level data and all results from the groundwater quality monitoring program were studied in detail. Three multilevel piezometers were installed at locations where it was felt that additional information was most required for a better interpretation (Figure 1, Appendix A). Samples were collected from the new piezometers and analyzed for major ions concentrations and dissolved organic carbon. Samples from on-site and off-site wells were analyzed for tritium.

Regional setting

Physiography, drainage and climate

The Oak Ridges Moraine, a prominent geomorphologic feature parallel to the north shore of Lake Ontario (Figure 1), is the divide between Lake Ontario and Georgian Bay-Lake Simcoe. The base of the moraine is defined by the 275 m.a.s.l. (metres above sea level) elevation contour, with the highest elevation reaching 405 m.a.s.l.. The hummocky terrain surface of the Oak Ridges Moraine is typical of an inter-lobate moraine. The topography is marked by extensive irregular slopes and steep depressions many of which contain lakes. Local relief is about 15 m.

Site 4 is located just south of the east-west trending line of kettle depressions along the crest of the moraine. Surface drainage is poorly developed in the relatively flat central part of the moraine where surface sediments consist of permeable sands and coarse silts. Beyond this, the moraine slopes gently to the Peterborough drumlin field in the northeast and the Schomberg clay plain in the northwest. Topography to the south of the moraine crest is very hummocky owing to knobs and kettles. Several of the depressions contain small lakes, ponds or swamps. The Markham-Pickering till plain lies beyond the south flank.

Site 4 is located about 2 km north of the boundary of the Holland-Black River basin which drains 1045 km² of the north flank of the moraine into Lake Simcoe. An average annual streamflow of 23 cm (or 29% of the average annu-

al precipitations) and average annnual baseflow of about 8 cm (10% of the pre-cipitations) are measured in the Holland-Black River basin (Vallery et al, 1982).

Information on climate in the site area has been recorded at the Oak Ridges climatic station. Over the period 1941-1970 the average daily tempera-ture varied from -7.3° Celsius in January to 20.6° C in July. The average total annual precipitation was 81.8 cm (CRA, 1985).

Geology and depositional history

In the study area, the Ordovician shale bedrock is overlain by a thick series of glacial tills and interglacial deposits. Overburden thickness usually exceeds 150 m and locally reaches 200 m. Horizons of glaciolacustrine silts and clays are present in the interglacial or ice-contact sand and gravel deposits. The regional stratigraphy is complex, with most stratigraphic units being irregu-lar and discontinuous.

Ice advancing south from the Lake Simcoe basin deposited a group of till and glaciolacustrine units found at the base of the Oak Ridges Moraine. The sand and gravel outwash forming the Oak Ridges Aquifer was deposited by meltwaters flowing west to southwest during ice retreat to the north. A subse-quent glacial advance from the Lake Ontario basin generated the fine grained Halton Till which generally covered the southern flank of the Oak Ridges Moraine. The northern extent of this ice advance was the crest of the moraine, in the vicinity of Musselman Lake (Duckworth, 1979). Blocks of ice left behind by the glacier retreating to the south formed kettle lakes and depressions. Meltwaters flowing northwest deposited sand and gravels to the north of the site. Kaye (1986), Kaye and Farvolden (1988) provided details.

Hydrogeology

The Oak Ridges Aquifer complex covers approximately 375 km^2 and consists of permeable sand and gravels interlayered with less permeable silts, clays, and tills associated with the kame moraine deposits. A series of low permeability tills and glaciolacustrine deposits are found at the bottom of the complex. Total thickness of the moraine complex locally exceeds 60 m and is on average 30 m, with the more permeable layers being considerably less in places.

The Halton Till overlies the aquifer complex to the south and as far north as the northern edge of Site 4 (Figure 2). A perched water table aquifer is found locally in this unit of low permeability.

The location of the regional groundwater divide is to the north of Musselman Lake (CRA, 1985). Flow is generally to the south, ultimately discharging in flowing wells or as springs or baseflow of streams 2 km to the south of Site 4. The horizontal gradient is regionally on the order of 0.012. A regional water table elevation map is included in Kaye and Farvolden (1988).

Under the Stouffville Landfill site and perhaps elsewhere along the Oak Ridges Moraine, the aquifer is unconfined. At the site, an unsaturated zone up to 20 m thick has been identified in the sand below the Halton Till. This unsaturated zone pinches out rapidly off the crest of the moraine to the south where the Halton Till behaves as a confining bed (Figure 3).

Based on hydraulic conditivity values, average annual infiltration varies from 11.5 to 14 cm/yr through the Halton Till to 46 cm/yr in gravel pits (CRA, 1985). Yields of domestic wells are generally good to excellent, as high as 45 l/s.

The groundwater of the Oak Ridges Aquifer is generally of excellent quality with a TDS of about 500 mg/l. The main ions in the groundwater, along with the approximate concentrations normally found here are: bicarbonate (360 mg/l); calcium (90 mg/l); sulphate ($<$ 30 mg/l); sodium (5-10 mg/l); nitrate (1 mg/l); and chloride (3 mg/l). These values are characteristic of modern recharge waters in sedimentary environments where dissolution of calcite and dolomite by milkly acidic infiltration waters is the main hydrogeochemical process (Howard and Beck, 1986).

Site description

Physiography and drainage

The natural topography at Site 4 was rolling and marked by 2 to 9 m deep kettle holes. Elevations varied between a high of 352 m in the west central area to a low of 321 m in the north. The upland area of the northern half of the site sloped steeply into the adjacent lowlands (Figure 4a). The kettles acted as surface water collection points draining most of the site (MOE, 1982a). The main route for the remaining runoff was to the northern swampy lowlands. The accumulation of wastes up to 17 m thick above the original ground surface over 34 ha disrupted the drainage pattern, causing surface drainage to the west of Highway 48 and to the south. Since the mid 70's, runoff has been redirected by a ditch and sewer system to a sedimentation pond in the northeast corner of the site (Figure 4b).

Waste disposal activities

Waste disposal operations began around 1962. Solid municipal waste, solid industrial waste and liquid industrial waste were received until 1970. During that period, solid waste was co-disposed with the liquid waste in the southwest corner of the site. The kettles were used for liquid waste disposal.

Liquid industrial waste received at the site included oily waste from service stations; clean-up waste from the food industry; hydrochloric acid; sulphuric acid; calcium hydroxide; sludges from water pollution control plants; and

waste from unknown sources. Until 1966, liquid waste was disposed of mainly in the south-central part of the site and in Lagoon 1 (MOE, 1982a). By 1970, five lagoons (Lagoons 1, 2, 3, 4, and 6) had been used to dispose of several thousand cubic metres of liquid waste, with the greatest actvity being in the interval 1966-1970. A sixth lagoon (Lagoon 5) received overflow (Figure 4a).

On air photos from 1954, Lagoons 2 and 3 contained water while Lagoon 4 and 6 did not, thus, indicating that the latter drained more quickly (CRA, 1985). Surface drainage of contaminated runoff in the southwest corner of the site, from Lagoon 5, and to east from Lagoon 1 was mentioned by early opera-tors. Liquid waste disposal ceased in 1970. By 1974, the lagoons were filled with solid waste and covered with earth. From then until closure in July 1985, only solid industrial waste and domestic waste were received at the site.

At the time of closure, approximatively 10 million tons of waste had been received at the site. A 90 cm layer of silty till and a 10 cm cover of topsoil have been installed on the site to limit infiltration and leachate produc-tion, and for cosmetic purposes.

Leachate chemical composition and distribution in the subsurface at the site became part of the hydrologic study (CRA, 1985). Test drilling was done at each lagoon and in the solid waste zones outside the lagoons. Soil samples were collected under the bottom of Lagoons 2 and 4 and analysed for EPA (U.S. Environmental Protection Agency) priority pollutants. Phenols, phtalate esters, polynuclear aromatic hydrocarbons and dichlorobenzene were detected in the soil samples. No pesticides or PCBs were detected.

During drilling in the waste now filling the natural depression at the site of Lagoon 5, an oily layer about 0.3 m thick was found at the elevation of the original water level in the lagoon. MOE analyzed aqueous and oil samples and several chlorinated hydrocarbons were detected. PCBs up to 6900 ng/L were found in aqueous unfiltered samples and up to 160 ng/L in filtered samples. In the oil samples, a maximum level of 3290 ug/g was found. A study of aquifer wells for all types of EPA priority pollutants concluded that there was no evidence of off-site contamination by the industrial organics present in the landfill (MOE, 1982b).

The migration of organic pollutants in the subsurface is retarded by sorption on organic material and soil clay-size particles. The soil samples were analyzed for their fraction of organic carbon (f.o.c.) and the percentage of clay-size particles. Soil f.o.c. values ranging between 0.01% and 0.11% were measured; these values, however, included the organic carbon from the pollutants. The fraction of material of grain size smaller than 50 um ranged between 17 and 32%.

Hydrogeology of the site

Surface water

The largest surface water body of the area, Musselman Lake, occupies a kettle depression (Figure 2). The only surface water connection to this lake is an intermittent stream that, when the stage is high, drains from the lake into the lowland area north of the site. This lowland extends to the west, becoming the head waters of the East Branch of the Holland River. A stream gauge from the site to 15 km downstream indicates low flow less than 0.028 m^3/s (Vallery et al, 1982). Kaye and Farvolden (1988) describe the relationship of Musselman Lake to the groundwater regime. Musselman Lake is a recharge zone for the Oak Ridges Aquifer, thus exerting a strong influence on flow in the aquifer. A shallow aquifer to the north of the lake provides some discharge to the lake which accounts for the stability of the lake level. The swamp that occupies the lowland extending west of Musselman Lake (Figure 2) apparently has a similar hydrologic regime. At the landfill site, this swamp receives discharge from the water table aquifer to the north and recharges the Oak Ridges Aquifer along the south side (CRA, 1985; Kaye and Farvolden, 1988).

Hydrostratigraphy

Three main hydrogeologic units are recognized at Site 4 (Figure 3). These units identified by CRA (1985) are: the Halton Till, a sandy silt till deposited during the latest stages of the Wisconsinan; the Oak Ridges Aquifer, composed primarily of outwash sands; and an aquitard at the base of the flow system at the site, comprised of three different stratigraphic units of glacial origin, all dense with low permeabilities.

The Halton Till is thin and discontinuous in the north part of the site, though reaching a 20 m thickness at the west site boundary. The composition of this sediment varies from a very clayey to a very sandy silt diamict.

The hydraulic conductivity of this unit was estimated by two different methods. Values of $1x10^{-2}$ to $3x10^{-7}$ cm/s were obtained from grain size analyses using the Hazen method (CRA, 1985). Hydraulic conductivity estimation of 10^{-6} cm/s in clay till and of 10^{-3} cm/s in the most sandy horizons were obtained by response testing (HCL, 1970).

The internal stratigraphy of the Halton Till at the site is very complex and areally variable. Local perched water table systems are found associated with the impermeable material, although at several places, the Halton Till is unsaturated throughout suggesting high permeability. Fast pathways to the aquifer through the till unit will be discussed in more detail later.

The Oak Ridges Aquifer is exposed in a narrow band along the northern part of the site, while overlain by the Halton Till to the south. Total saturated aquifer thickness increases from 34 m in the north of the site to 44 m near the south boundary. The unsaturated zone increases from a few metres thickness in the north to almost 20 m in the southeast corner.

Duckworth (1979) provided a detailed description of the aquifer sediments from exposures near the site and identified two major units. The lower unit consists of medium and coarse sand, massive or in crossbedded sets with interbeds of cross laminated fine sand. The overlying unit consists of subparallel beds of gravel up to 250 m long, 20 m wide, and 12 m high, dipping to the southwest, enclosed in up to 100 m of beds of laminated fine sand and silt. Gravel also occurs as small longitudinal bar deposits. Logs of boreholes drilled at the site describe units similar to the gravel filled crevasses in sequences of silty sand described by Duckworth (1979).

Even on the site scale, stratigraphic correlations within the aquifer are very difficult because of the limited extent of units, even the identifiable clay and gravel horizons. Aquifer material is generally silty with varying clay content. The sediments of the lower half of the aquifer are mostly poorly sorted fine silty sands. In the upper half, sediments ranging from clay lenses to gravel beds are found. A local clay unit is present in the aquifer in the southwest corner where coarse gravelly sands are more abundant than elsewhere in the upper half. Silty horizons are more frequent near the water table in the northeast corner area.

Hydraulic conductivity estimates from grain size analyses of aquifer material with the Hazen method range between 2×10^{-1} and 1×10^{-6} cm/s (CRA, 1985). Response testing in the aquifer lead to an hydraulic conductivity estimate of 10^{-3} cm/s (HCL, 1970).

Dense, fine grained deposits acting as aquitards underlie the aquifer below the site and as many as three distinct stratigraphic units may be repre-

sented. The first is a silty sand till with interbedded horizons of cemented sand and gravel. A clay and silt unit of glaciolacustrine origin is found below the till. The lowest unit identified at Site 4 is a clayey silt till with hydraulic conductivity on the order of 10^{-8} cm/s. For practical purposes, these deposits simply represent the bottom of the flow system.

Hydrology of the aquifer

Flow in the aquifer under the site is generally to the southwest and horizontal hydraulic gradients as low as 0.004 are observed. Recharge from Musselman Lake; the outwash material between the lake and the site; and the bog area north of Site 4 contribute to the maintainence of a stable groundwater flux.

Recharge from the sedimentation pond in the northeast corner affects the water table configuration. A recharge mound can be drawn if the water levels of shallow wells of this locality are interpreted as representing hydraulic heads in the main aquifer and so are included in the contours (Figure 5a). If mounding does occur, it would have the most influence during the spring, disrupting the normal flow pattern and producing off-site gradients. In the winter, mounding would decay and have less influence on the flow system.

Conversely, if the water elevation in these wells of the sedimentation pond area is considered as perched water in the silty horizons just above the water table, the effect of the recharge basin on the main aquifer flow pattern is much smaller (Figure 5b). There is much evidence which suggests that the water levels measured here actually indicate a perched system. The water level

of the shallow wells of the recharge basin area varies seasonally. Discharge to the north from this locality does not seem to occur. The gradients necessary for such flow are not observed and downward flow is indicated by an aquifer piezometer nest in the slope from the site to the swamp to the north (Figure 6).

Since 1981, the mean water level of some wells in the southwest corner have been rising steadily to gain almost 1 m. Changes in the surface drainage may have caused this; generally, however, seasonal water level variations in aquifer wells are 0.3 to 0.6 m. Highs usually occur in the spring and sometimes in the fall, near the sedimentation pond though elsewhere, the water table is rather stable.

Four major production wells are located in the vicinity of the site: two industrial wells between 1.5 and 3 km east of the site and two municipal wells 2.25 km south supplying the Town of Stouffville. A 24 hour pump test was performed on one of these (municipal well #5) in June 1965. A transmissivity of 62.4 m^2/day and a storativity of $2x10^{-2}$ were estimated from the pump test data (HCL, 1965). Observations of the static level of wells and open gravel pits in the area during the test indicated that pumping from the municipal wells does not have a major impact on the regional hydraulic potential distribution.

A radius of influence (based on 3 cm of drawdown at the outer edge) was calculated by CRA (1985) with an aquifer hydraulic conductivity of 10^{-1} cm/s. The 1230 m radius obtained represented a worst case situation since the aquifer is generally less permeable than assumed for the calculation, therefore, the gradients are steeper and the zone of influence less extensive.

Private wells for domestic supply were located along Highway #48 very close to the landfill site. These wells produced a total of about 10 l/s and were abandoned in the fall of 1983 when municipal piped water was made available.

Hydrology of the Halton Till

The material applied on waste as daily cover is rather impermeable. At some locations, perched water is the result of infiltration trapped by the daily cover. Wells completed within the refuse at Lagoons 1, 3, 5 and 6 and in the southwest corner, have never been reported to be dry. LW 7-81 and LW 3-81 (Figure 7) are consistently dry which suggests that the entire thickness of the refuse at Lagoons 2 and 4 is unsaturated at places.

Drillers' logs of holes penetrating the Halton Till show highly variable stratigraphy which results in different infiltration rates over the site. Some observations are indicative of generally impermeable conditions: thin unsaturated zone; numerous locations of ponding and perched water; very stable aquifer well hydrographs; and a good barometric efficiency in a deep aquifer well of the southwest corner. Conversely, the existence of some permeable areas is indicated by a minimal amount of runoff from the site before it was altered and the rapid infiltration of the liquid waste in the disposal lagoons. Also, in several localities, the Halton Till is unsaturated throughout the total unit, indicating good drainage and recharge conditions.

Perched water table conditions seem to occur in several small zones of limited extent. Wells or excavations encountering unsaturated soils around

these zones mark their maximum lateral extent. No perched water has been found in the zones along the south and east boundaries of the site. A minimal amount of water is observed in walls of excavations along these boundaries, draining small isolated pockets of saturation, and all shallow wells in those areas have been consistently dry. Northwest of Lagoon 5, a dry well (OW 2-81) (Figure 7) indicates a 10 m thick unsaturated zone in the upper part of the Halton Till.

Off-site gradients are present at some locations in the Halton Till. Along the west boundary, a perched water table (OW 1-82, 2-82) (Figure 7) drains off-site from mid-summer to the end of the fall, and east, towards the site, for the rest of the year. The perched water table is found at about 2.5 m below the surface. The maximum horizontal gradient to the west is 0.015 while to the east, it is 0.025 (CRA, 1985). There usually is a strong vertical component.

Flow components to the west are present within the perched system in the southwest corner but generally flow is downward.

The perched water table in the area of Lagoon 5 approximately follows the topography and shallow groundwater is found in the till at the bottom of the north slope (Figure 6), suggesting lateral groundwater flow to the north. Field examinations of the slope, however, have failed to identify spring discharge from the bottom of the refuse and a strong downward gradient is observed in piezometers at the bottom of the slope. The downward gradients and the absence of discharge along the slope suggest that flow through the Halton Till at this location is predominantly downward. Similar hydraulic condi-

tions prevail north of Lagoon 2 and 3 and in the southwest corner. It can be expected that flow in the perched water table of these areas is predominantly downward.

Permeable pathways to the aquifer might exist near the zones where no perched water is found in the Halton Till or the refuse. The hydrograph of some shallow piezometers show rapid rises followed by rapid drops in the water level. This indicates that water percolates downward through the more permeable zones in the waste. OW 12-83 in the southwest corner of the site (Figure 7) is sometimes dry and could indicate a window to the aquifer. The stratigraphic log of OW 12-83 describes very sandy material through a large part of the till thickness. Similarly, extreme water level fluctuations of 4.9 m in LW 10-81 (in the Lagoon 5 area) could also reflect a window to the main aquifer.

In general, observation well hydrographs in the perched water table suggest poor hydraulic connections within the local zones of perched groundwater in the Halton Till. Even if off-site gradients are indicated by some wells, the actual lateral migration is limited by the low permeability of the material. Theorically, it is expected that the Oak Ridges Aquifer would influence the entire system by acting as a drain and drawing the flow downwards. Field evidence has confirmed this condition.

Groundwater quality monitoring

Monitoring program

The source of contamination in the Oak Ridges Aquifer at Site 4 is waste and waste products burried in the Halton Till in the lagoons and in the refuse. Some wells in the perched water table apparently show background water quality. The absence of highly elevated indicator parameters in water samples from piezometers a few metres away from waste supports hydraulic evidence for mainly vertical flow in the Halton Till. Since migration of the contamination is mainly downward toward the water table in the aquifer, no attempt was made to characterize in detail migration rates and pathways through the the till and the unsaturated zone. Instead, this study has focussed on contaminant migration in the Oak Ridges Aquifer. The water table in the aquifer is the upper boundary of the system considered. The strength of the contamination at this boundary is monitored by water table wells.

The program of groundwater quality monitoring at Site 4 can be divided into two phases. In the first phase starting in 1968, groundwater quality in a limited number of on-site wells and private off-site wells was monitored quarterly to produced an extensive record of chloride, conductivity, hardness, sulphate and phenol levels.

The hydrologic investigation of the Site by CRA from 1982 to 1984 brought about a considerable extension of the network of aquifer monitoring wells. The new wells became part of the quarterly monitoring program and;

calcium, magnesium, sodium, potassium and alkalinity levels were also determined on only four occasions. Since 1985, the quarterly analyses include these parameters, along with pH and dissolved organic carbon (DOC). The second phase of the program included monitoring in all wells starting in 1982.

Water quality data from the monitoring program are "noisy". This noise could be caused by other factors than impact from the site. It might be caused by natural variations due to changes in hydraulic or chemical conditions or by spurious variations associated with sample collection and sample analysis. In phase one data, large shifts in water quality from background values due to contamination are recognized even if other sources of water quality variations affect the data. However, the noise renders the detection of trends difficult in the shorter record of water quality data of phase two; natural variations have to be accounted for to distinguish trends due to contamination.

Information on concentrations of organic pollutants in the aquifer was gathered by the MOE from November 1981 to October 1983. MOE sampled biweekly, analyzing for total and dissolved organochloride, pesticides and PCBs.

Contaminant plume assesments from phase one data

MOE assesment

The variations in levels of inorganic contaminants before 1982 were interpreted by MOE (1982a). Chloride and conductivity seemed most appropriate to assess plume migration because they are observed to be the least erratic of parameters with time. On-site and off-site wells of the southwest part of the site showed a rise in contaminant levels followed by a decrease or a plateau during the passage of a contaminant pulse in the area of those wells (Figure 8). The chloride arrival curve of some wells showed fluctuations that were thought to represent 3 or more pulses of contaminants (MOE, 1982a). In the Hutchinson well (Figure 7), the tail of the earliest pulse was observed in 1968 and the last peak occurring, in 1978, reached 80 mg/l. The arrival time of this last pulse front was observed in a series of 8 wells along the flow path and successively reached wells located farther downgradient indicating the migration of the contamination pulse. Breakthrough first occurred in OW 16-70 in 1976 and finally reached the most downgradient well of the monitoring array, OW 1-76 (Figure 7) in 1978-80 (MOE, 1982a).

The successive arrival in downgradient wells, from OW 16-70 to OW 1-76 was interpreted by MOE as the migration of a unique pulse of contaminants. Several hypotheses were advanced for the creation of this pulse. The first one was the filling of Lagoon 1 with wastes in 1973-74. This left the pulse a period of only 2-3 years to migrate through the sediments above the

main aquifer and then laterally to reach OW 16-70 (MOE, 1982a). Fast migration above the water table was explained by the presence of a window of more permeable material near field capacity under the former liquid disposal lagoon.

The origin at Lagoon 1 was first considered because of its position, directly upgradient from the impacted wells. If lateral migration in the Halton Till and/or dispersion of contaminants in the aquifer are very important, a pulse originating at Lagoons 2, 3, or 6 could be the cause of contamination in wells indirectly downgradient from these locations. The filling of the lagoons in 1973-74, could have generated a pulse; however this would have required horizontal migration with very high velocities on the order of 80 m/yr.

MOE (1982a) proposed another possible explanation for the origin of this contaminant pulse (Figure 8): rapid infiltration of the liquids disposed of in Lagoon 1, or in Lagoons 2, 3, and 6. MOE (1982a) considered that the most probable origin of the pulse was the filling of Lagoon 1 in 1973-74 (MOE, 1982a).

The arrival time at successive downgradient wells was used by MOE to obtain migration velocities of 122 to 564 m/y. Using these velocities, a constant porosity of 0.3 and the gradient between the wells led to estimates of hydraulic conductivity ranging from 1×10^{-2} to 1×10^{-1} cm/s. According to MOE (1982a), the consistent hydraulic conductivity values supported the idea of one pulse migrating under the southwest corner. Off-site migration of one pulse was, however, questionable since contamination reached the Bolton well and the Hutchinson well at the same time although they are at different distances downgradient. MOE (1982a) recognized many difficulties in the interpretation, mainly due to insufficient information.

CRA assesment

CRA (1985) reassessed the historical curves basing the interpretation on chloride values. According to CRA (1985), the record of on-site wells showed the passage of several pulses under the site while dispersion caused a smooth breakthrough curve at off-site wells. The migration of a unique pulse away from the site described by MOE (1982a) was rejected because of the discrepencies left without explanation.

Cause of the early contaminant rise is attributed to either the liquid waste disposal in lagoons, or to filling of the lagoons (CRA, 1985). The latter implies that permeable windows exist under the lagoons. It is concluded that the most probable origin of the pulses is the liquid waste disposal in the lagoons which ended in 1970. Pulses in the on-site wells are only the reflection of spatial and temporal variations in the liquid waste disposal activities (CRA, 1985).

Mounding under the lagoons during the period of active liquid waste disposal may have disturbed the flow field and created hydraulic gradients to the south. A general southwestern migration pattern has been observed, however, indicating that if mounding had occurred it then would have had little influence on contaminant migration.

Migration velocities of 103-1116 m/year were calculated from the chloride arrival times at different wells; these velocities, with a porosity of 0.3 and a gradient of 0.004, were used to determine hydraulic conductivity over the site (CRA, 1985). The values obtained range from $3x10^{-2}$ to $3x10^{-1}$ cm/s.

In both the MOE and CRA assessments, the successive arrival times of contamination at downgradient locations were used to calculate the plume migration velocity. The arrival time was considered to correspond to the first increase in the level of chloride in the observation wells. This approach neglected the dispersion taking place at the fringe of the plume and overestimated the velocity. The hydraulic conductivities determined were very high and not compatible with later assessment of the plume.

Review and analysis of phase one results

Increase in levels of at least two out of four parameters monitored occurred in the observation wells designated 16-70, 7-73, 1,2-75, 1,2-76, Bolton, and Hutchinson (Figure 7). During the same monitoring period, natural water quality conditions were recorded at the Baker, Deacon and Hoover wells.

In OW 16-70 and OW 7-73, chloride levels began to rise around 1976 and started to decrease in 1981 (Appendix A), indicating that a zone of elevated chloride had migrated downgradient from these wells. In the southwest corner, chloride concentrations started to increase in OW 2-75 and 2-76 around 1980 and about a year later in the Hutchinson and Bolton wells (Appendix A), suggesting that a zone of elevated chloride was in the vicinity of OW 2-75 and approching the private off-site wells.

MOE and CRA interpreted the phase one curves as showing the migration of contamination from Lagoon 1 to the off-site wells of the southwest corner at very high velocity. In this study, the curves are explained by the slow migration of two different zones of elevated chloride, one taking origin at Lagoon 1, and a second from Lagoon 4.

An increase in hardness and sulphate is observed at the impacted wells at the same time as the increase in chloride. However, sulphate concentrations began to decrease at OW 16-70 and OW 7-73 in 1978, when chloride concentrations had remained more or less constant. Also in wells of the southwest corner, sulphate started to decrease when chloride concentrations were still increasing. If the migration of sulphate was not affected by an attenuation process, sulphate levels would be increasing in the wells downgradient, as observed for chloride. Decreasing levels of sulphate, both on and off-site, suggested that the sulphate concentration has been lowered by a process acting on and off-site. Hardness levels were generally stable, suggesting no downgradient migration of calcium.

Evolution of the plume 1982-1987

Trends in background concentrations.

Water quality trends due to impact from Site 4 are less obvious in phase two data than in phase one data and are difficult to distinguish from changes due to natural causes. In order to attribute a trend at a downgradient location to contamination, it should be shown that the trend is not caused by natural variations. Assuming that seasonal variations affect upgradient and downgradient data equally, the water quality data in wells located upgradient are used to determine if trends due to natural causes are affecting the data from downgradient wells.

Trends in the concentration of the major ions in the upgradient wells between 1982 and 1987 were studied in two groups of wells: three wells located

northeast of the site and three wells west of the site (Appendix B). The upgradient wells showed an increase in chloride concentration which reached a maximum in 1986 and then decreased. Since this trend seemed to be general, chloride variations at more wells were examined. Also upgradient from the plume, although in the center area of the landfill, OW 36-83b and OW 37-83a indicated the same trend (Figure 9). The trend was also observed at the onsite downgradient wells of the south and southwest. The trend was not present in the wells where the highest chloride concentrations were observed. However, in shallower wells of that area, the same gradual increase in concentration until 1986 was observed. Off site, concentrations in the Bolton well showed a discernable peak in 1986 while this was not observed in the Hutchinson well.

In summary, the increase in chloride concentration in 1986 was observed in deep and shallow wells over the site except downgradient in the wells located near the core of the plume. This trend in chloride concentrations has to be considered in the analysis of the chloride plume development.

Background levels of alkalinity, calcium, sodium, magnesium, potassium and sulphate from 1982 to 1987 were variable and did not indicate a trend. Over the same interval, magnesium concentrations showed a peak in 1986 at all the northeast wells. No trend was noticed to the west, where concentrations of magnesium were generally lower. In general, no significant temporal trends were observed, except for chloride.

Contour maps.

Contour maps of chloride, sulphate, alkalinity, sodium, calcium, magnesi-

um and potassium at different moments were prepared to study the spatial evolution of the plume with time. Average values over a year were used to minimize the effect of seasonal variations and of errors related to sampling and analytical procedures. Averages are often used to ease the detection of trends in very variable data, since averages are always subject to smaller variations than the individual observations (Sgambat et al, 1981). The three years chosen were 1982, 1984 and 1986. The wells installed in 1987 were sampled once and the results were included in the 1986 maps. Means and standard deviations were calculated for unaffected conditions. A contamination threshold equal to about the average plus one standard deviation was used as the value of the lowest contour on the maps. The maximum values at multilevel piezometers were used to obtain, as much as possible, a plan view through the core of the plume.

Contour maps of chloride variations using the difference between downgradient and average upgradient concentration were prepared to eliminate the effect of the background chloride temporal variation. In 1982, a chloride plume extended off the southwest corner of the site, and the highest values (42 and 48 mg/l above background) were found there and in the center of the south landfill boundary. High chloride levels were found at two locations near the bottom of the aquifer in 1984. The peak concentrations of 122 mg/l in the southwest and 91 mg/l above background in the center could be contoured as the core of two small plumes as shown on Figure 10 (all other contour maps are found in Appendix C). By 1986, the two peak values had decreased to 71

and 82 mg/l above background. An additionnal zone of high chloride concentration was found near the bottom of the aquifer, downgradient from the center plume. Four small plumes could be contoured in the envelope. The plume area increased from 1982 to 1986, expending downgradient towards the west and the south in the center part, but not to the southwest. From 1984 to 1986, the highest values had decreased and the contours of lower concentration had spread, indicating that the plume had migrated and dispersed (Figure 10). Over that period, the center of the plume as moved about 70 m downgradient.

Sulphate plumes for 1982 were contoured in the southwest corner and in the center area. By 1986, the southwest plume had decreased in size and maximum concentration; the center plume had extended donwgradient. A plume to the west contoured for 1984 had migrated downgradient by 1986.

Two calcium plumes were contoured in the southwest and the center for 1982. The center plume decreased in size in 1984 but increased again in 1986. The core of the southwest plume migrated downgradient from 1982 to 1986.

Only one small magnesium plume was observed in the southwest corner until 1987, when three small plumes were defined by contouring. The new plumes were located in the center, and in the west downgradient from Lagoon 5.

Three zones of high potassium concentration were contoured in the southwest corner, in the center and downgradient from Lagoon 5 in 1982. This last zone had disappeared in 1984 and the two other plumes had moved downgradient. In 1986, the monitoring netword was extended westward and the western plume could be contoured again. The center and southwest plumes were contoured downgradient.

Three sodium peaks were recognized downgradient in 1982, two in the center, and one in the southwest. In 1984, one center peak had disappeared. In 1987, the core was directly downgradient from the 1982 location but the other center peak had disappeared. Two plumes very similar in shape to the chloride plumes can be contoured with the maximums at the same location. Many isolated peaks were present. The two small zones of higher alkalinity levels contoured in 1982 changed in shape with time and extended downgradient.

Values for DOC have been available only since 1985. High values were found in the center and the southwest. A plume could be contoured northwest of Lagoon 5.

The monitoring network was extended during the four year period described. The additional information available from new wells increased the accuracy of the delineation. Chloride was used to determine the extent of contamination and to delineate in some detail subregions of the large plume (Figure 10). Sodium was also useful for these purposes and showed the same trend as chloride. The other parameters were only slightly higher than background. DOC was higher than background downgradient from the site, and also higher than background to the west of the site.

For most parameters, two zones of elevated values are contoured: in the center and the southwest corner. The first location is downgradient from Lagoons 1, 2, 3, and 6 and the second location is more directly downgradient from Lagoon 4. A third zone with elevated potassium, magnesium and calcium has been identified downgradient from Lagoon 5. Isolated high values of sodi-

um and chloride were found in 1987 in that area but low values were recorded at wells directly upgradient before 1987. The contour maps for the three periods indicate a downgradient migration of all major ions.

Plume delineation in cross sections

In a stratified aquifer, the distribution of contaminants in cross section also has to be considered in detail. Levels of the different parameters for a limited number of locations in the aquifer are available for that purpose.

Several parameters were contoured in two aquifer cross sections (Figure 11) along the flow path and bisecting the plumes defined in the broader envelope. Wells were projected onto the cross sections. All contours were drawn by hand with the values found in Appendix C. Aquifer stratigraphy in the sections was drawn from borehole logs and Duckworth's (1979) description of the structures found in the fluvioglacial deposits forming the aquifer (Figure 12).

In cross section A-A' large plumes of chloride (Figure 13), conductivity, sulphate and sodium are delineated. They are located approximately at the same distance downgradient except for the sodium plume located farther downgradient. DOC is high at only one location which corresponds approximately to the position of the chloride, conductivity and sulphate plumes center of mass. Calcium, magnesium and potassium plumes are small and located upgradient from the other plumes. The plumes are lenticular and have a downward orientation suggesting a sinking action at the front of the plume. Sodium and sulphate plumes are more horizontal in the upgradient part. The plumes show a subdivision in two zones at the downgradient end.

In cross section B-B' also, plumes of chloride (Figure 13), conductivity and sulphate are larger than the other plumes, and extend farther downgradient. Size and downgradient extent of the sodium and DOC plumes are more limited. For most parameters, a larger plume in the upper half of the aquifer and a smaller one near the aquifer bottom are contoured. The plumes of the two zones are horizontal. The contours show two independent plumes but the values could also be contoured showing a link in the middle part. The stratigraphy directly upgradient from the two plumes is uncertain since no boreholes were drilled at that location. Very highly permeable sediments in the form of a crevasse filling of limited horizontal extent have been noted elsewhere in the aquifer. The presence of both a shallow and a deep plume in cross section B-B' could be explained by the division of a plume due to such heterogeneities in the aquifer at that location. However, such heterogeneities were not included in the interpretation shown on Figure 12.

The phase one data for OW 1,2-75 and OW 1,2-76 show an earlier arrival in the deepest piezometers with highest levels recorded in the shallow wells indicating the migration of two pulses, the deep one having a higher velocity than the shallow one. During a natural gradient test at the Canadian Force Base, Borden, Ontario (Sudicky et al, 1983) the tracer zone split into two segments travelling at different velocities because of local heterogeneity. Similarly, the chloride breakthrough curves of the southwest corner could be explained by the migration of the two segments from an original slug fragmented because of spatial variations in hydraulic conductivity.

The two cross sections intersect different lagoons. A-A' crosses Lagoon 1, 2, 3, and 6, while B-B' is close to Lagoon 4. The presence of several highs within the plumes of cross section A-A' might be explained by the use of disposal lagoons at different times creating different slugs.

Influence of dispersion from field description

At the Borden site, Ontario, a wide fan-shaped plume developed under the influence of a transient water table mound causing components of flow transverse to the general direction of flow (MacFarlane et al, 1982). Long and thin plumes were observed at the Long Island sites studied by Kimmel and Braids (1980) where the water table slope was uniform and relatively stable with time.

The chloride plume delineated at Site 4 falls between these two extremes. The plume is more or less elliptic, and about the width of the landfill. The cores have a similar shape and are sligthly wider than the lagoons. The presence of concentration irregularities inside the plume is related to the multiple sources rather than to a low degree of mixing. Mounding at the water table during the years of liquid waste disposal might have caused transverse spreading of the plume. The evolution of the plume over the period 1982-1986, however, reflects migration in a rather stable regime with moderate transverse dispersion.

Field evidence of weak values of vertical dispersivity in horizontally migrating plumes in glaciofluvial aquifers are presented by MacFarlane et al (1983) and Kimmel and Braids (1980). The chloride plumes at Site 4 are relatively thin in cross section which is consistent with these findings.

Hydrogeochemical characterization of the plume

Natural water quality

Groundwater in the Oak Ridges Aquifer in the Stouffville area generally has a TDS lower than 500 mg/l. The water is of the Ca^{2+}-HCO_3^- type with low chloride and sulphate concentration in about the same proportion (Howard and Beck, 1986). Water quality is typical of recharge in material of significant carbonate content. Saturation indices calculated with the speciation program WATSPEC (Wigley, 1977) indicate that the water is saturated with respect to calcite and undersaturated with respect to gypsum.

Groundwater quality downgradient of the landfill is affected by the waste disposal activities. All parameters listed in Table 1 (except pH) are found downgradient at levels significantly above the average upgradient level. However, the maximum concentrations in the plumes are at least one order of magnitude lower than the highest concentration in the leachate. The maximum concentrations of chloride and sulphate downgradient are much lower than the recommended drinking-water standards. Volatile organics, base neutral and acid extractable organics, pesticides and PCBs have been monitored by CRA at six on-site wells since 1985. No pesticides, PCBs or dioxins were measured above the detection limit. No organic compounds were found at concentrations above background concentrations. Volatile organic compounds were reported at low concentrations in two downgradient wells (OW 30-83b and OW 12-84a). DOC concentrations are low (< 20 mg/l). A rapid decrease in DOC concentrations

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from a landfill to downgradient wells can be explained by degradation of the organic compounds and dilution of the leachate (Baedecker and Back, 1979) or fixation on the sediments above the water table.

Hydrochemical processes in the plume

The hydrochemical processes occurring in the plume were studied to explain the relative position of the plumes of different ions. Water quality in wells of the east section of the plume corresponding to the downgradient half of cross section A-A' was used for these purposes. The variation of sodium, calcium, sulphate and chloride along the flow path was examined (Figure 14a).

The variation of Cl⁻ along the flow path suggests that two small plumes followed each other. The Na^+ plume is behind the most downgradient Cl⁻ plume but in front of the second one. Ca^{2+} is high behind the Cl⁻ plume but further downgradient; it follows the same trend than Cl⁻. SO_4^{2-} increases slightly along the flow path.

Saturation indices were calculated to verify the thermodynamic potential for dissolution or precipitation of the most common minerals containing Ca^{2+} or SO_4^{2-}; calcite and gypsum. The results indicate a slight supersaturation with respect to calcite and undersaturation with respect to gypsum at all locations. Precipitation of calcite is thus possible while dissolution of gypsum could occur if the mineral was present.

The selectivity coefficient K_s of a cation exchange reaction such as:

$$A^+ + B_{sorbed} \longrightarrow B^+ + A_{sorbed}$$
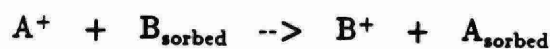
is expressed as:

$$K_s = [B^+](A_{sorbed}) / [A^+](B_{sorbed})$$

and such an equilibrium relationship can be written between all cations present in groundwater. If the proportions of the different cations in the groundwater change, cation exchange takes place to adjust to the new conditions. Usually, Ca^{2+} is the main cation on the exchange sites since it is the dominant cation in the groundwater. It is sorbed preferentially to other cations. Loading of other cations in the water by contamination often causes the elution of Ca^{2+} from the exchange sites (hardness halo) that is replaced by other cations. At Site 4, Ca^{2+} is the main cation in uncontaminated groundwater (Table 2). In the contaminated groundwater, Na^+ is the cation showing the largest concentration contrast with background conditions. Sorption, precipitation/dissolution and Ca^{2+}-Na^+ exchange are the most probable causes of variations in water quality that are not directly due to mixing of contaminated water with the original groundwater.

Variations in concentrations along the flow path could be caused by source strength variations, hydrodynamic dispersion and chemical reactions. To distinguish variations due to dispersion from variations caused by chemical reactions, ratios of the conservative species chloride to the other species were calculated and plotted versus distance along the flow path. An increase in the ratio could mean that attenuation is taking place and a decrease in the ratio could indicate an addition of the species to the water.

The validity of the interpretation of chemical processes from chloride ratios depends on the validity of the assumptions required for their use: a constant and homogeneous Cl^-/species ratio in the leachate and the use of points along a flow line for comparison (Nicholson et al, 1982). The chloride concen-

tration in the point sources is variable across the site (Table 1). However, it is relatively stable with time at each location. The variations in chloride concentration at the source limit the accuracy of the interpretation from Cl^-/ species ratios.

The ratio curves are described in three events, referred to as event I, II, and III (Figure 14b). In the core of the plume, corresponding to event III of the ratio curves, the change in the proportions of Ca^{2+} and Na^+ in the groundwater (Table 2) could explain the addition of Ca^{2+} to the water and the diminution in the Na^+ concentration observed. After the passage of the core of the plume, in event II, conditions become closer to background thereby causing the population on the exchange sites to adjust again through the reaction opposite to the one occurring in the core. Na^+ is added to the groundwater and Ca^{2+} is taken out of solution indicating that Ca^{2+} is replacing Na^+ on the exchange sites. In the most upgradient part of the plume, event I, Na^+ is attenuated by another process other than cation exchange with Ca^{2+}, maybe simple sorption since Ca^{2+} is also taken out of solution. The Ca^{2+} decrease might be caused by calcite precipitation or sorption.

The Cl^-/SO_4^{2-} ratio indicates an attenuation of SO_4^{2-} followed by an addition in event II, and attenuation again in event III. The SO_4^{2-} attenuation is not a result of gypsum precipitation; it may be due to bacterial sulphate reduction as indicated by the H_2S odors at some wells (CRA, 1985). The addition of SO_4^{2-} to the water in event II could be the result of gypsum dissolution if the mineral is present in that part of the aquifer.

The relative position of the plumes of different parameters is influenced by these chemical reactions taking place in the aquifer to adjust to the higher concentrations. The Cl$^-$/species ratios indicate that attenuation of Na$^+$ is occurring upgradient of the contoured Na$^+$ plume. The Ca^{2+} plume is retarded by cation exchange with Na$^+$. Downgradient, the Na$^+$ plume is retarded by cation exchange with Ca^{2+}. The increase in Ca^{2+} concentration, however, does not result in a noticeable hardness halo since the concentrations are still within the range of natural variations. The small SO$_4^{2-}$ plumes are the result of attenuation in front of and behind the center of the plume.

Use of environmental tritium

Tritium is a radioactive isotope of hydrogen with a half-life of 12.43 years. The abundance of tritium is expressed in tritium units (1 TU = 1 ^{3}H/10^{18} hydrogen atoms). Before the start of nuclear testing in 1953, background levels were around 10 to 15 TU (Egboka and al, 1983). Radioactive decay over the ensuing 34 years would result in levels below 2 TU today.

Water samples were collected from 21 wells in May 1987 for tritium analysis. The samples were analyzed at the Environmental Isotopes Laboratory of the University of Waterloo by direct liquid scintillation counting with a precision of $\pm$ 8 or by gas counting after sample enrichment with a precision of $\pm$ 5 TU. Values measured in the aquifer ranged from 0.8 to 73.5 TU. The values were of the same order of magnitude as values mentioned by Howard (1986) for the Oak Ridges Aquifer. Howard and Beck (1986) stated that in the Oak Ridges Aquifer, even under the cover of Halton Till, recent CaHCO$_3$

waters were extensive and had tritium concentrations ranging from 23 to 132 TU.

The samples analyzed for tritium were collected in wells screened at different levels in the thick aquifer. The distribution of the results in the aquifer cross sections A-A' and B-B' are shown in Figure 15.

To determine of groundwater age zones, the flow paths in the aquifer and the change in concentration of tritium in recharge with time are required. Flow in the aquifer is generally horizontal, except under the site, as shown by the contaminant plume configuration. The concentration in recharge is assumed to be approximately equal to the concentration in precipitations. Tritium concentration in precipitations in the Stouffville area (Figure 16) is obtained by weighting the tritium rainout in Ottawa with annual rainfall at the Town of Angus and the Toronto airport. This input is affected by radioactive decay. The resulting values in 1987 are also shown on Figure 16. The migration of tritium in fluvioglacial aquifers is also largely influenced by dispersion (Egboka and al, 1983) which spreads and lowers the tritium input.

A concentration of only 0.8 TU in a sample from the deep part of the aquifer indicates that water more than 34 years old is present near the bottom of the aquifer. Samples with less than 15-20 TU when analyzed by direct counting commonly have no significant tritium if analyzed again by enriched counting (Egboka et al, 1983). The concentrations outside the plume delineated with chloride and in the deepest two thirds of the aquifer are lower than 25 TU. The groundwater of these zones might also represent pre-1953 recharge.

Inside the contoured chloride plume, concentrations as high as 74 TU were measured in samples from the shallow depths while lower values were obtained for deeper zones (Figure 15). Groundwater representing recharge after 1970 had, in 1987, tritium levels lower than 70 TU (Figure 16). Concentrations of 65-74 TU in groundwater from shallow wells indicated that the water in the plume recharged the aquifer before 1970. From 1963 to 1970, the tritium concentration in precipitations was decreasing. If water in the plume represented recharge before 1970, the most recent recharge, with relatively low tritium content, should be found closer to the water table. Older recharge, with higher concentrations, should be found at greater depth. The reverse situation was observed at Site 4 which was not compatible with a plume that having for origin leachate derived from precipitations.

The distribution of ^{3}H concentrations in the plume could be explained if the origin of the plume is liquid waste free of bomb tritium disposed in the lagoons. Dilution of the input from precipitation with the liquid waste could result in low tritium concentrations of the older part of the plume. Deep in the aquifer, the lower values inside the plume could represent liquid wastes mixed mostly with pre-1953 water.

Flow modelling

The plume was analyzed by means of numerical flow model in a cross section to provide a more quantitative forecast of future plume development. An aspect of the flow system of particular interest at Site 4 is the nature of recharge and its influence on the plume. This aspect was investigated by comparing the flow patterns under different recharge conditions to the observed plume. The influence of the heterogeneity of the aquifer on the flow patterns was also studied.

Model

The model used is the finite-element dual formulation of flow described by Frind and Matanga (1985). The formulation of the equation governing flow in terms of potential and stream function allows reduction of numerical errors and easier detection of errors in the simulated flow field. The stream functions correspond to flow lines and allow a better visualization of the flow paths. The solution of the dual formulation of flow is influenced by the hydraulic conductivity distribution in the domain, and by the boundary fluxes. Flow in the aquifer is more or less uniform and steady, allowing application of such a model.

Cross section A-A' oriented parallel to the direction of flow was chosen for simulations using this technique. It was assumed that flow in or out of the section (of unit thickness) is absent, limiting the problem to two dimensions.

The program FLONET originally written in FORTRAN by E.O. Frind and modified by E.A. Sudicky of the Institute for Groundwater Research of the University of Waterloo was used for the simulations. The grid having 1056 triangular elements provided sufficient resolution to allow a reasonable description of the heterogeneity of the aquifer material. No information was available on the degree of anisotropy of the medium. It was considered isotropic.

The stratigraphy in the cross section was established using drillers logs, sample studies and borehole resistivity logs. Descriptions of the structure of the granular deposits forming the aquifer were used for the correlation of individual well data. An average hydraulic conductivity value was calculated for each different lithology present in the cross section from CRA (1985) data. The hydraulic conductivity values that were used are listed in Table 3.

Boundary conditions

The boundary conditions of the model (Figure 17) were chosen according to field data. The northern end was represented by a flux (q_1, q_2, and q_3) and the bottom by an impermeable boundary. The southwestern end was represented by constant potentials: 33.59 m above a low permeability layer and 33.08 below. Field data were not available for the early years of liquid waste disposal at the site. In the simulation of these early conditions, it was assumed that the same boundary conditions could be used, except at the water table. The effect of increased recharge in the lagoons during liquid waste disposal was not likely to have reached the southwest boundaries of the system, located more than 700 m downgradient from the lagoons. It was calculated with the Jacob's modification of the Theis nonequilibrium well equation:

$$s = \frac{0.183\,Q}{T}\log\frac{2.25\,Tt}{r^2 S}$$

using values of transmissivity (T) and storage coefficient (S) derived from pump test data (HCL, 1965) that injection in the aquifer at a rate of 36000 m³/yr (Q) for ten years (t) would result in only 50 cm increase in the water level (s) at the boundary (r).

A flux boundary was used at the water table. The position of the water table, a function of the system, was solved for iteratively. The potential solution was then compared to the observed water table. The wells located close to A-A' were used to draw the observed water table. In the recharge area, several shallow wells screened in silty sands indicated hydraulic heads 1.5 m above the hydraulic heads measured in piezometers screened at a lower level in the aquifer. Hydrographs of these wells recorded large seasonal hydraulic head variations (up to 4 m). Hydrographs of deeper piezometers of the same area showed much more stability. Hydraulic heads measured in shallower wells were not considered representative of aquifer conditions. Water could be perched in the silty sediments above the water table and misinterpreted as part of the water table. Only the wells with rather stable hydrograph or screened at a level clearly below the water table were used for the model.

Calibration

The model was first calibrated by adjusting the fluxes at the water table and at the northeast boundary until a good match with the water table as observed in 1986 was obtained. The final values of the fluxes at the northern

boundary q_1, q_2, and q_3, were respectively 0.005 m/yr, 0.001 m/yr, and 0 m/yr. A recharge rate corresponding to the average precipitation in the area (about 80 cm/yr) was used at the sedimentation pond of the northeast corner. Since 1985, the landfill has been capped by a 91 cm layer of silt till and 10 cm of topsoil. Percolation rate through solid waste under these conditions was estimated to be about 13 cm/yr (CRA, 1985) and this value was used for the recharge flux under the landfill. The water table was best simulated without a flux component downgradient of the landfill, suggesting no contribution of recharge to dilution of contaminants at that location. Groundwater throughflow from recharge upgradient of the landfill, however, was noted to be available for contaminant dilution. Under these conditions, a shallow plume would develop in the aquifer (Figure 18a).

The influence of the heterogeneous medium on the flow field can be seen by comparing the flow patterns obtained when heterogeneity is taken into account with the homogeneous case (Figure 18b). Flow in the heterogeneous medium is more complex. After a zone of downward flow under the recharge locations, the flow is directed upward by diversion around zones of lower hydraulic conductivity. Flow is relatively horizontal farther downgradient where the material is more homogeneous. At the end of the section, flowlines go around zones of lower hydraulic conductivity in the middle of the aquifer. Flow is downward, towards the more permeable underlying sediments. Plumes delineated in cross section A-A' show an orientation towards the bottom of the aquifer at the end of the cross section. The heterogeneity in hydraulic conductivity has to be taken into account to obtain flowpaths corresponding to the plume configuration.

Very small upward or downward hydraulic gradients were measured (CRA, 1985) in the zone where horizontal flow is predicted by the model, and larger downward gradients were measured at the downgradient end. Unfortunately, there were no wells in the portion of the flow domain where the model predicted upward gradients. In general, the flow field representation of present conditions corresponded to the field observations.

Simulations

The effect of other recharge conditions on the flow patterns was simulated by varying the water table recharge function, while keeping the value of all other parameters fixed. The conditions corresponding to two periods of the landfill history were simulated: the period of liquid waste disposal (1962-1970) and the period following until the site was capped (1970-1985). The only difference between the recharge conditions of the two periods was the higher recharge under the lagoons in the first period.

The first period recharge function consists of recharge in the northeast corner, under the landfill, in lagoons (Figure 17). Recharge in the lagoons along A-A' only was considered: in Lagoons 1 and 2 located at about the same distance and in Lagoon 6 farther downgradient. In the early years, the sedimentation pond of the northeast corner was not developed but historically, drainage was reported to occur in that area. The same infiltration rate was used as for present conditions. Under the landfill, a recharge corresponding to 50% of the precipitations was used, as obtained from tritium measurements at the Borden site (Egboka et al, 1982).

Infiltration rates in the lagoons during the time of liquid waste disposal is unknown. It was estimated by the operator, however, that before 1966, volumes of liquid waste on the order of 250 m³ were received each month (MOE, 1982a). An infiltration rate of 0.2 m/yr was obtained by dividing this volume by the area of the lagoons in use at that time (Lagoon 1, 2, 3 and 6). This estimate was considered to be too low, especially considering that the lagoons were used to a greater degree after 1966. The sensitivity of the solution with respect to that parameter was tested. Rates of 1 m/yr in Lagoons 1 and 2, and of 2.5 m/yr in Lagoon 6 were used as a first possible situation. By doubling the recharge rates in the lagoons, a 22% increase of the maximum horizontal velocity was obtained. However, the flow lines had a similar configuration.

Recharge under the lagoons influences the simulated flow field by increasing the downward component of flow under the lagoons. If concentration of inorganic contaminants is higher in the liquid waste than in the leachate from the solid waste, by advection only, a plume with two inside cores would result (Figure 19a) similar to the plumes delineated in cross section A-A'. The greater density of the highly contaminated water relative to natural groundwater may have contributed to the downward migration of contaminants under the lagoons. The effect of the density contrast is difficult to estimate. The relatively low concentrations observed in the plume today suggest that dispersion diminished largely this effect.

The flow patterns resulting from the simulation of period (1970-1985) are very similar to those obtained for the previous period (Figure 19b), although

the flux of water in the aquifer is much lower. By advection, a large uniform leachate plume would develop.

The northern boundary condition is a specified flux and the material is less permeable in that part of the aquifer. Consequently, simulation of early recharge conditions caused a rise of the water table at that boundary (Figure 20) resulting in a larger gradient and horizontal groundwater velocity.

Horizontal velocity calculated by the model for present conditions ranged from 0 to 90 m/y and averaged 13 m/y. For early conditions with a recharge rate of 1 m/y in Lagoons 1 and 2, and 2.5 m/y in Lagoon 6, horizontal velocities ranged from 0 to 159 m/y with an average of 22 m/y. For the period 1970-1985, horizontal velocities up to 123 m/yr were obtained with an average of 17 m/yr. Very low velocity were calculated by the model at the bottom of the aquifer and upgradient of the recharge area, where tritium results indicated that the water is more than 34 years old. The highest velocities calculated were found under the recharge locations, close to the water table, and in the lower part of the aquifer at the downgradient end. In general the values obtained decreased toward the bottom of the aquifer.

The distribution of velocity in the domain could be used with a finite elements transport model. An estimate of the average linear groundwater velocity after integration over the variations is obtained in the following chapter.

Transport modelling

Chloride breakthrough curves are available for the phase one wells. These curves are the result of advective-dispersive solute transport over several hundred metres. The parameters controlling migration at that scale can be estimated from the breakthrough curves by a fitting procedure with a transport model.

In two dimensions the equation governing the transport of conservative species in groundwater can be written as:

$$D_z \frac{\partial^2 c}{\partial z^2} + D_z \frac{\partial^2 c}{\partial z^2} - \bar{v} = \frac{\partial c}{\partial t}$$

where c is the concentration, t is the time, x and z represent the distance from the source in respectively the longitudinal and transverse direction, and $\bar{v}$ is the average longitudinal groundwater velocity. D_z and D_z, the coefficient of longitudinal and transverse dispersion are defined as:

$$D_z = \alpha_z \bar{v} + D_d$$

$$D_z = \alpha_z \bar{v} + D_d$$

where α_z is the longitudinal dispersivity, α_z the transverse dispersivity, and D_d is the coefficient of molecular diffusion.

The solution of the 2D transport equation for the following initial and boundary conditions:

$$c(z, z, 0) = 0$$

$$c(\infty, z, t) = 0$$

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$$c\,(-\infty,\ z,\ t) = 0$$

$$c\,(0,\ z = z_1\ to\ z_2,\ t) = c_0$$

can be written as:

$$\frac{c}{c_0} = (z_2 - z_1)\left[erfc(\frac{z - \bar{v}t}{2\sqrt{D_z t}}) - exp(\frac{z\bar{v}}{D_z})erfc(\frac{z + \bar{v}t}{2\sqrt{D_z t}}) \right]$$

$$+ \frac{1}{\pi} exp(\frac{z\bar{v}}{2D_z})\sum_{n=1}^{\infty}\frac{1}{n}\cos(\frac{n\pi z}{b})\left[\sin(\frac{n\pi z_2}{b}) - \sin(\frac{n\pi z_1}{b}) \right]$$

$$X\ \frac{\bar{v}b}{A}\left[exp(\frac{-zA}{2bD_z})erfc(\frac{zb\ At}{2b\sqrt{D_z t}}) - exp(\frac{zA}{2bD_z})erfc(\frac{zb + At}{2b\sqrt{D_z t}}) \right]$$

where:

$$A = \sqrt{v^2 b^2 + 4D_z D_z n^2 \pi^2}$$

This solution was programmed by E. A. Sudicky at the University of Waterloo for the injection of a slug.

For the curve fitting, a source of constant concentration active from 1966 to 1970 was used, corresponding to the period of most intense liquid waste disposal. It was assumed that the migration of the liquid waste from the lagoons downward to the water table in material near saturation took less than a year. The amount of time taken by the center of mass of the four year slug to travel from the source to an observation well (arrival time) depended on the average groundwater velocity only. The first step was to adjust the velocity to match the center-of-mass arrival time observed on breakthrough curves. The dispersivity was then adjusted to fit the entire curve. The analytical solution fit the chloride measurements relatively well (Figure 21).

Final values obtained for velocity and dispersivity are listed in Table 4. Longitudinal dispersivities obtained from calibration of models on observed

plumes of non-reactive contaminants in different sand and gravel deposits range between 12 and 200 m, and are estimated to about 21 m in glacial deposits (Anderson, 1979). The values obtained at Site 4 compare very well with the dispersivity obtained for glacial deposits.

A ratio ($\frac{\alpha_L}{\alpha_T} = 10$) much lower than those reported in the literature was used to match the breakthrough curves. Anderson (1979) cited a value of 300 and Frind and Hokkanen (1987) obtained a valued of 1000.

Several assumptions were used to simulate the breakthrough curves; the values of velocity and dispersivity obtained are only rough estimates. However, the values were consistent over the site which would support the scenarios suggested. Using these values and assuming that the source has become inactive in 1970, the model predicted that in about 10 years from now, the chloride levels will be back to background levels (Figure 20). The chloride plume configuration, however, suggested that leachate from the solid waste has started to reach the aquifer. The plume resulting from contamination by leachate should migrate downgradient at a velocity on the order of those obtained with the transport model. The displacement of the center of the plumes delineated in 1984 and 1986 indicated a migration rate of about 35 m/yr which was compatible with that forecast.

Summary and conclusions

Contaminants have migrated downward through the Halton Till and reached the main aquifer resulting in the off-site migration of a plume of contaminated groundwater downgradient to the southwest. Contaminant levels in observation wells showed several peaks over the history of monitoring of the site. The peaks were attributed to dumping of liquid waste in the lagoons that drained quickly, introducing large volumes of contaminants to the aquifer directly upgradient from the observation wells.

Contaminants move downward with infiltrating water where sandy material increases the bulk permeability of the Halton Till. Lateral migration in the till is very slow and apparently is not a serious threat. Detailed hydrogeologic analyses show that little or no lateral off-site migration of contaminants occurs within the Halton Till. The swamp to the north of Site 4 is not threatened by contaminated groundwater discharge from the landfill since it is a groundwater recharge zone, marked by strong downward gradients. Surface water in the vicinity of Site 4 is, therefore, protected by the natural groundwater flow system.

The contaminant plume was defined by an elevated chloride concentration (20 mg/l) and originated from leachate of municipal solid wastes in the landfill, which is still taking place. The liquid wastes, draining quickly in lagoons, caused the elevated zones of major ions now downgradient from the lagoons within the main plume. In plan view, the plume is elliptic as a result

of dispersion while the spread of contamination through most of the aquifer thickness was caused by heterogeneities in the stratigraphy.

Recharge conditions under the site had a major influence on the plume development. The high infiltration rates in the lagoons increased the contaminants migration velocity. Available groundwater throughflow from recharge upgradient of the site diluted the contaminant concentrations in the plume. In the cross section studied, recharge downgradient of the site does not seem to contribute to dilution, however, it might be more important at other locations.

The transport model predicts that concentration in the plume caused by liquid waste will continue to decrease. Most of the contamination due to solid waste in the landfill, however, has yet to impact the aquifer. The configuration of major ion plumes suggests that chloride has started to reach the aquifer while other major ions are still retained in the Halton Till. The maximum concentration in the future plumes is unknown. The strength of the contamination as it enters the aquifer can be considered as the source of the plumes. Monitoring of the on site water table wells should be performed to follow the evolution of this source. According to results flow model results, the extent of this plume resulting from leachate infiltration at low rates will be limited to a shallow portion of the aquifer. The predicted migration velocity of the plume is on the order of 35 m/yr.

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Table 1: Water quality

	Leachate range at Site 4		Upgradient average	Downgradient maximum		Standard
Cl	120	3500	11	82		250
Ca	370	4600	80	157		
DO	580	2360	4.6	20		
K	29	585	1.1	2.6		
pH	5.69	12.37	7.6	6.59	10.59	
Na	334	680	4.5	139		
SO4	10	1650	23	84		250
Mg	0.1	202	12.7	26		

Table 2: Cation exchange

	Concentrations mmol/l				Proportions			
	Ca	Na	Mg	K	Ca	Na	Mg	K
Background	2.	.22	.5	.025	80	10	20	1
Downgradient	2.5	.87	.5	.05	50	20	10	1

Note: Average concentration at OW 16-70 in 1986 as
downgradient concentration

Table 3: Hydraulic conductivity values

Material	K (cm/s)
Clayey silt	3.3×10^{-6}
Sandy silt	2.0×10^{-5}
Fine sand	1.4×10^{-4}
Medium sand	6.5×10^{-3}
Coarse sand	8.0×10^{-3}
Gravel	2.0×10^{-2}

Table 4: Transport model results

well	v (m/yr)	α_L (m)	α_T (m)
OW 7-73	27	20	2
OW 2-76	16	15	2
OW 16-70	20	18	2
BOLTON	38	40	5
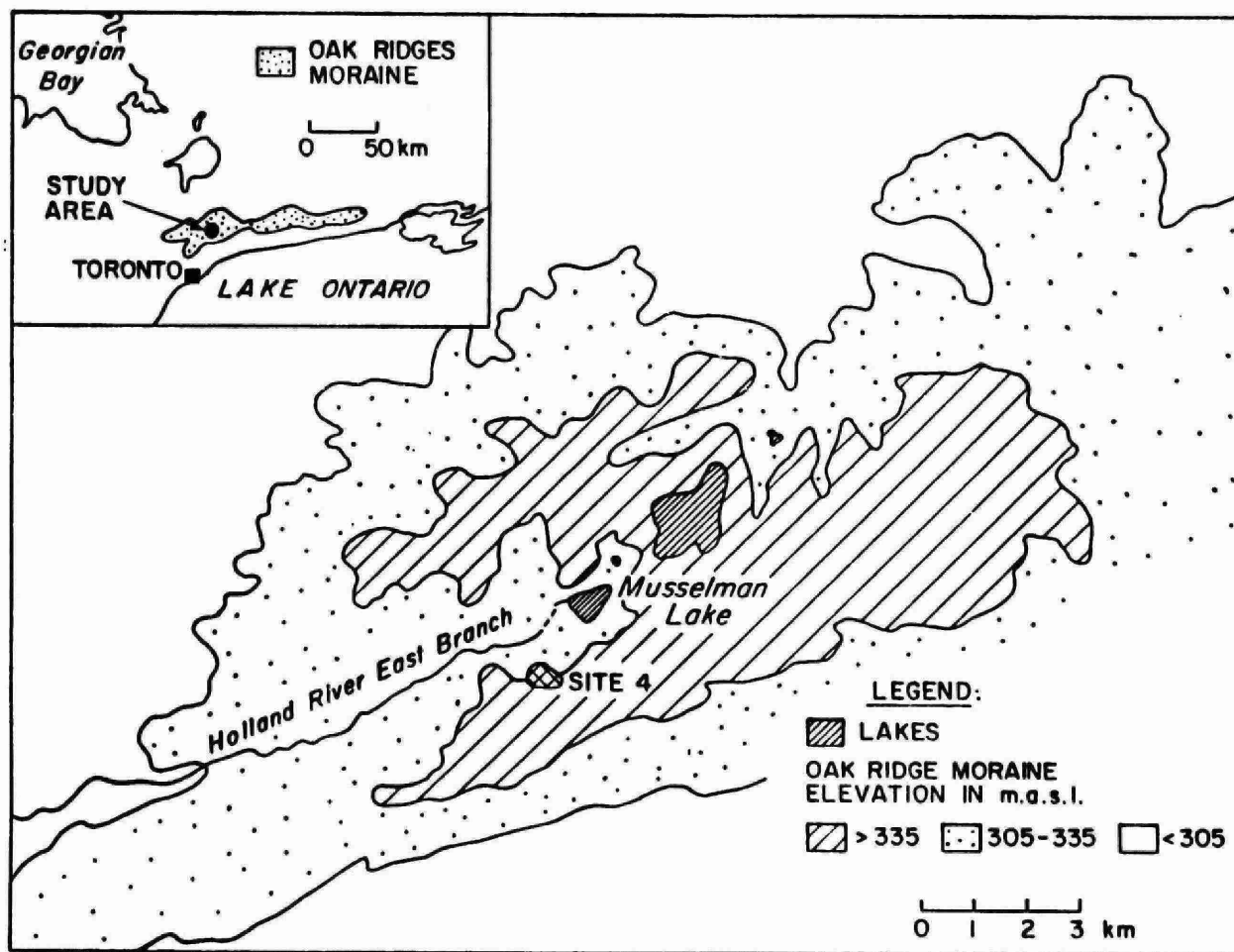
Figure 1:   Regional setting
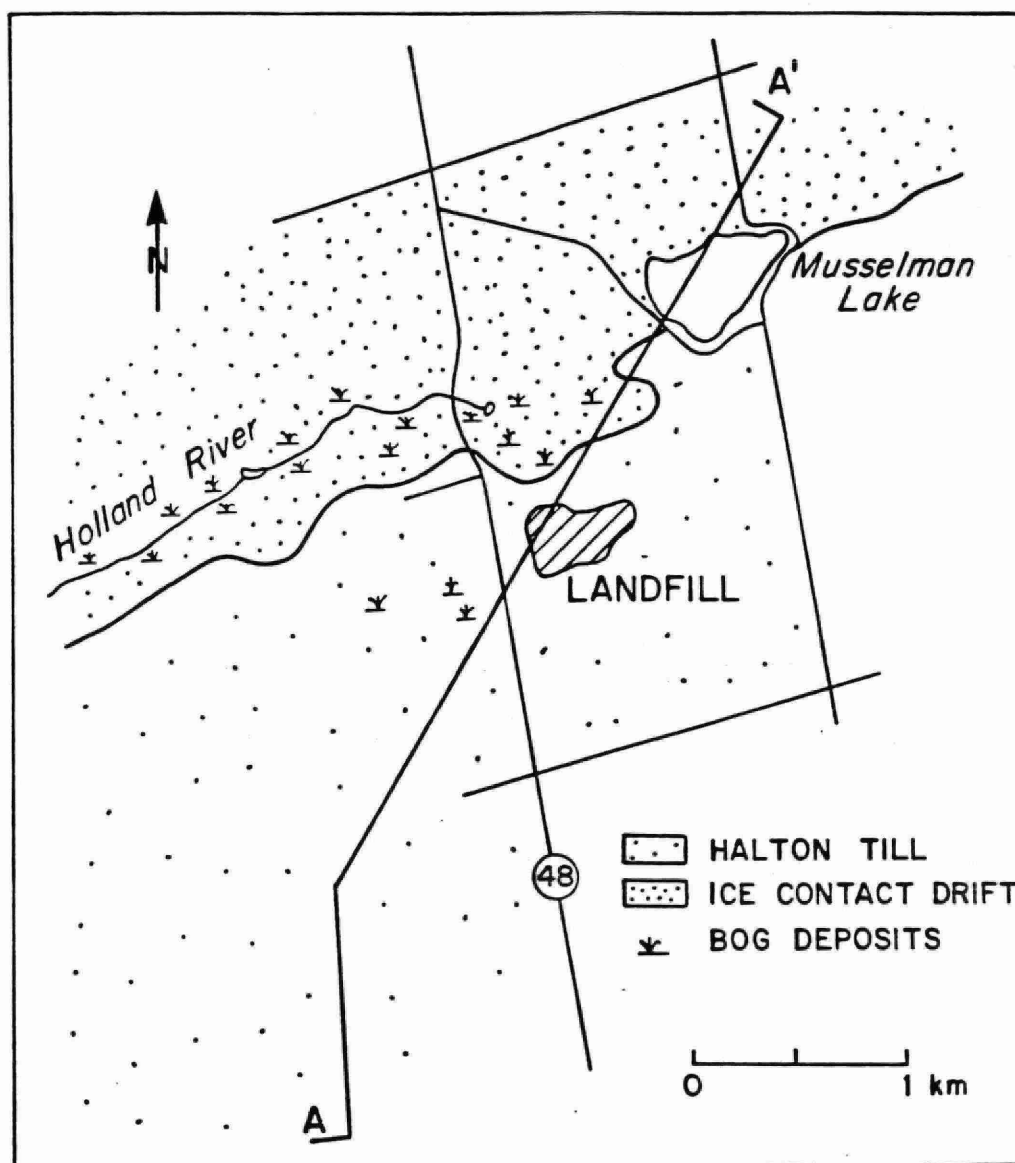
Figure 2:   Surface sediments
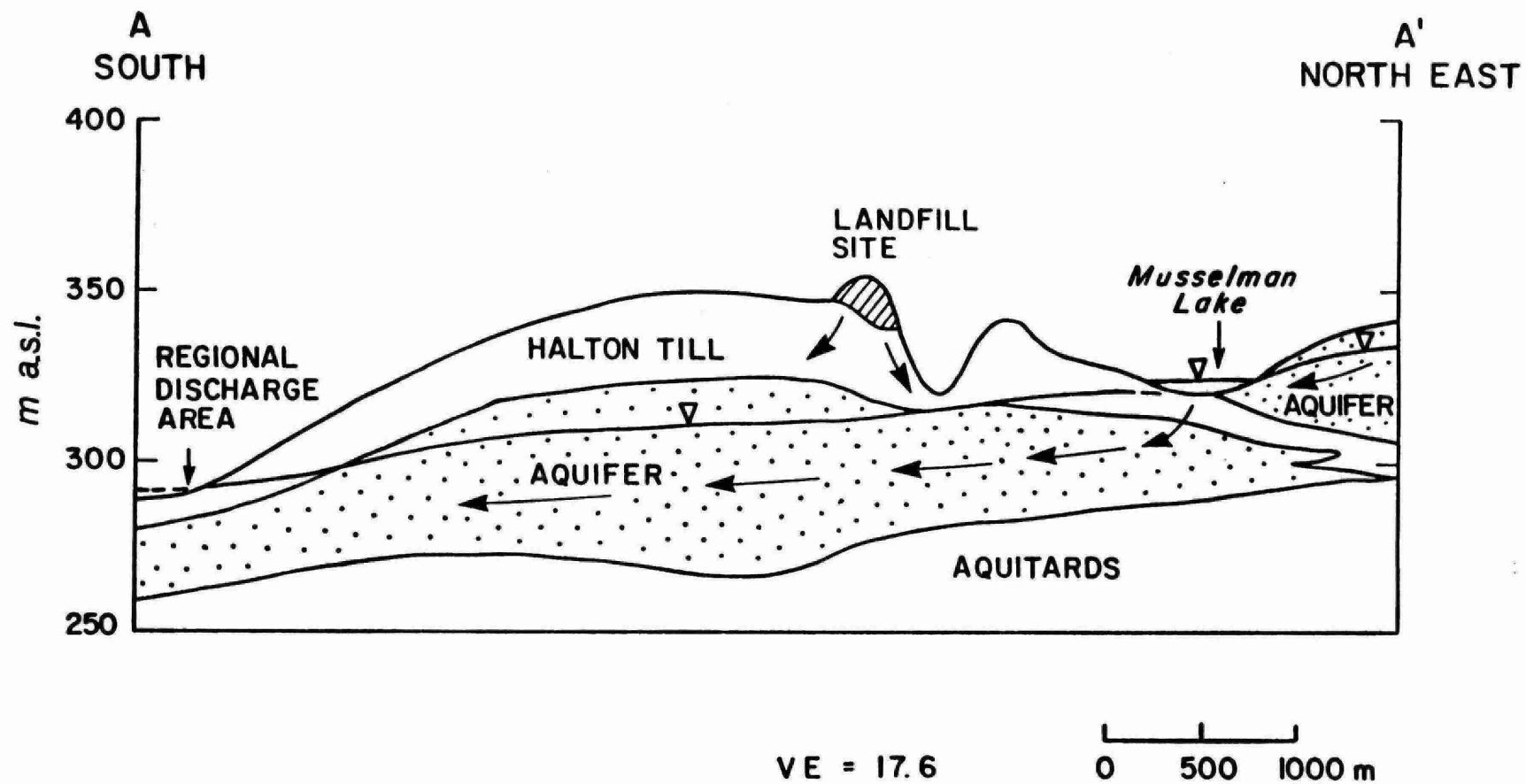
Figure 3:   Hydrostratigraphy and flow system
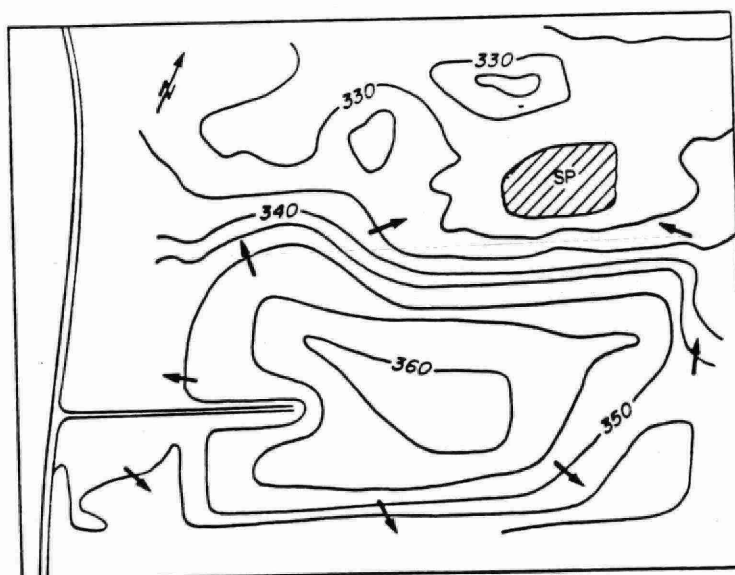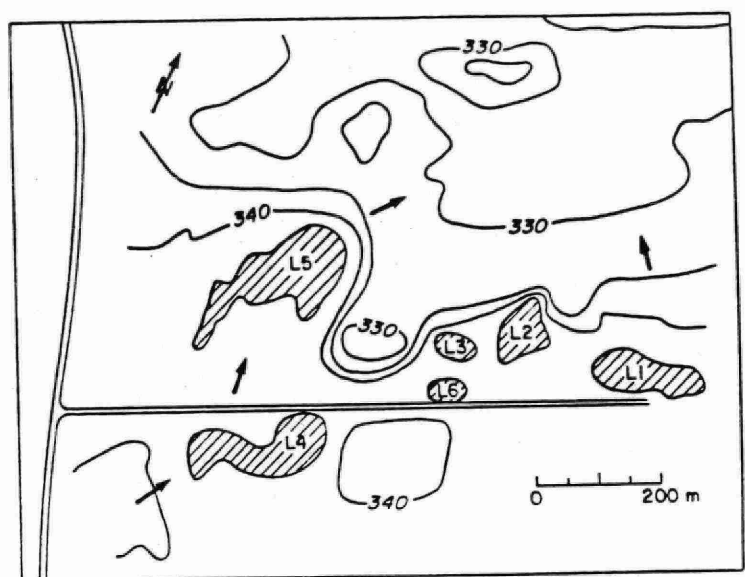
LEGEND:

L3   LIQUID WASTE DISPOSAL LAGOON

SP   SEDIMENTATION POND

→   DIRECTION OF SURFACE RUNOFF

⌐360⌐   ELEVATION IN m.a.s.l.

Figure 4:   Topography and drainage at Site 4.   a) Early operation conditions b) At time of closure
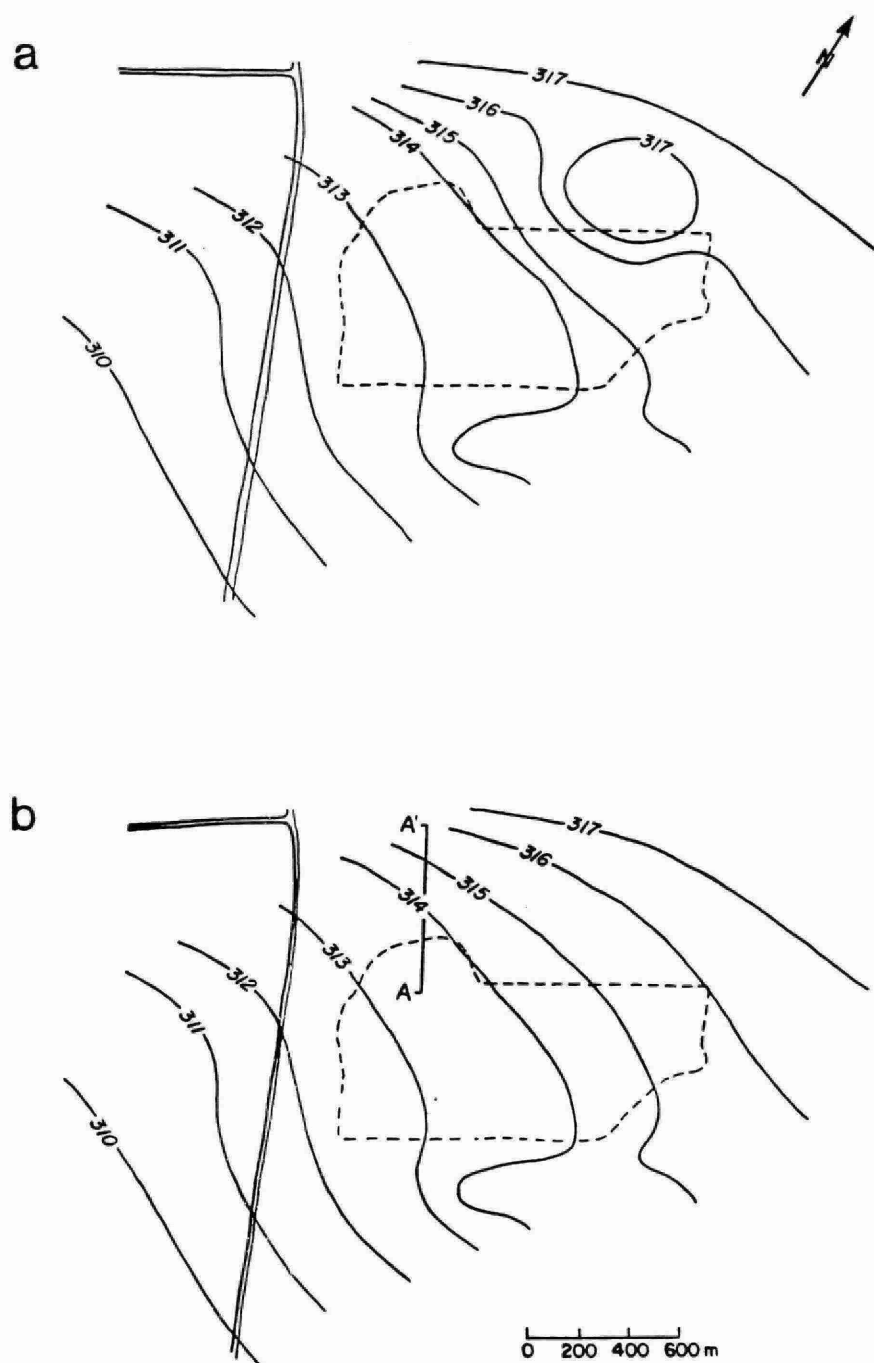
Figure 5: Coutour maps of the head in the main aquifer in m.a.s.l..
a) Including a mound b) Without a mound
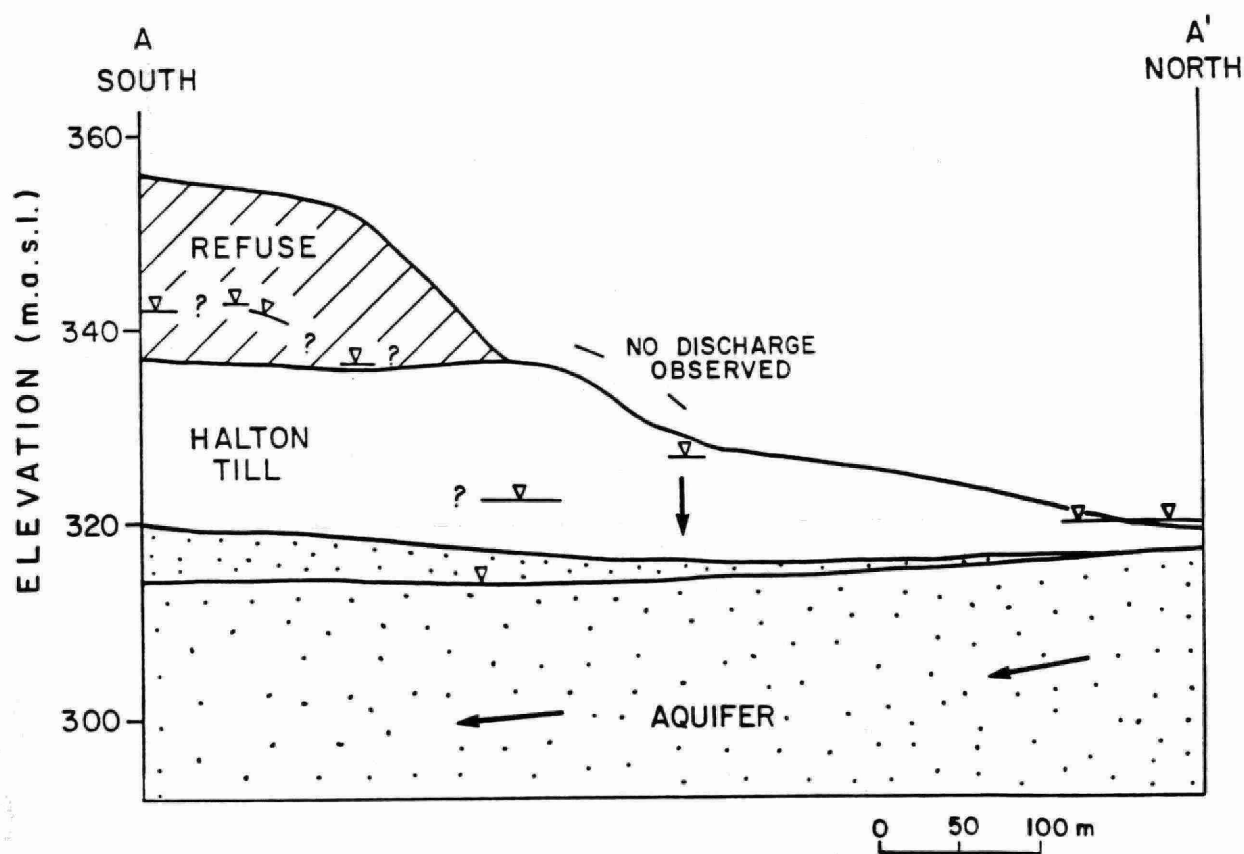
Figure 6:   Cross section of the north slope
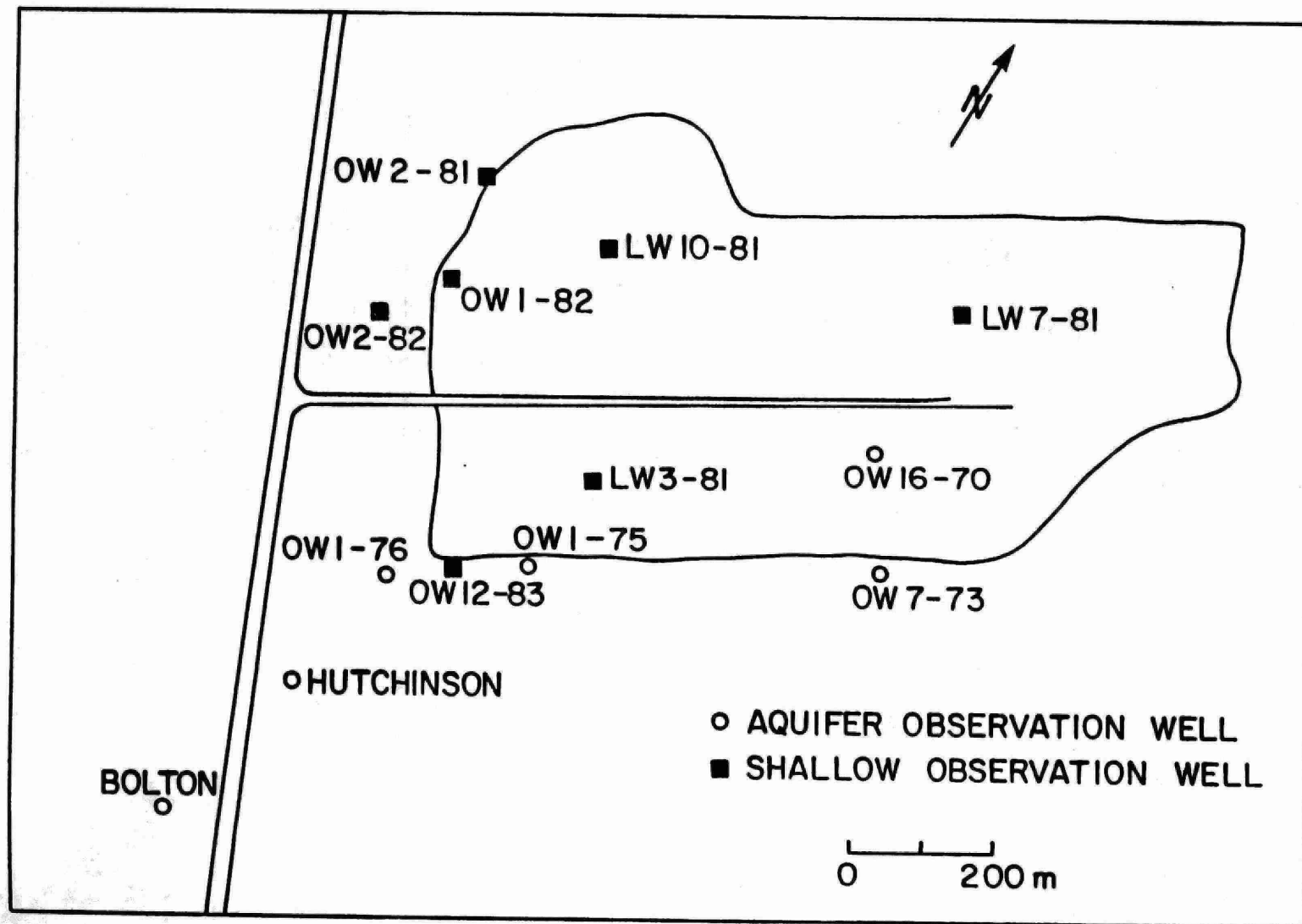
Figure 7: Observation wells location map
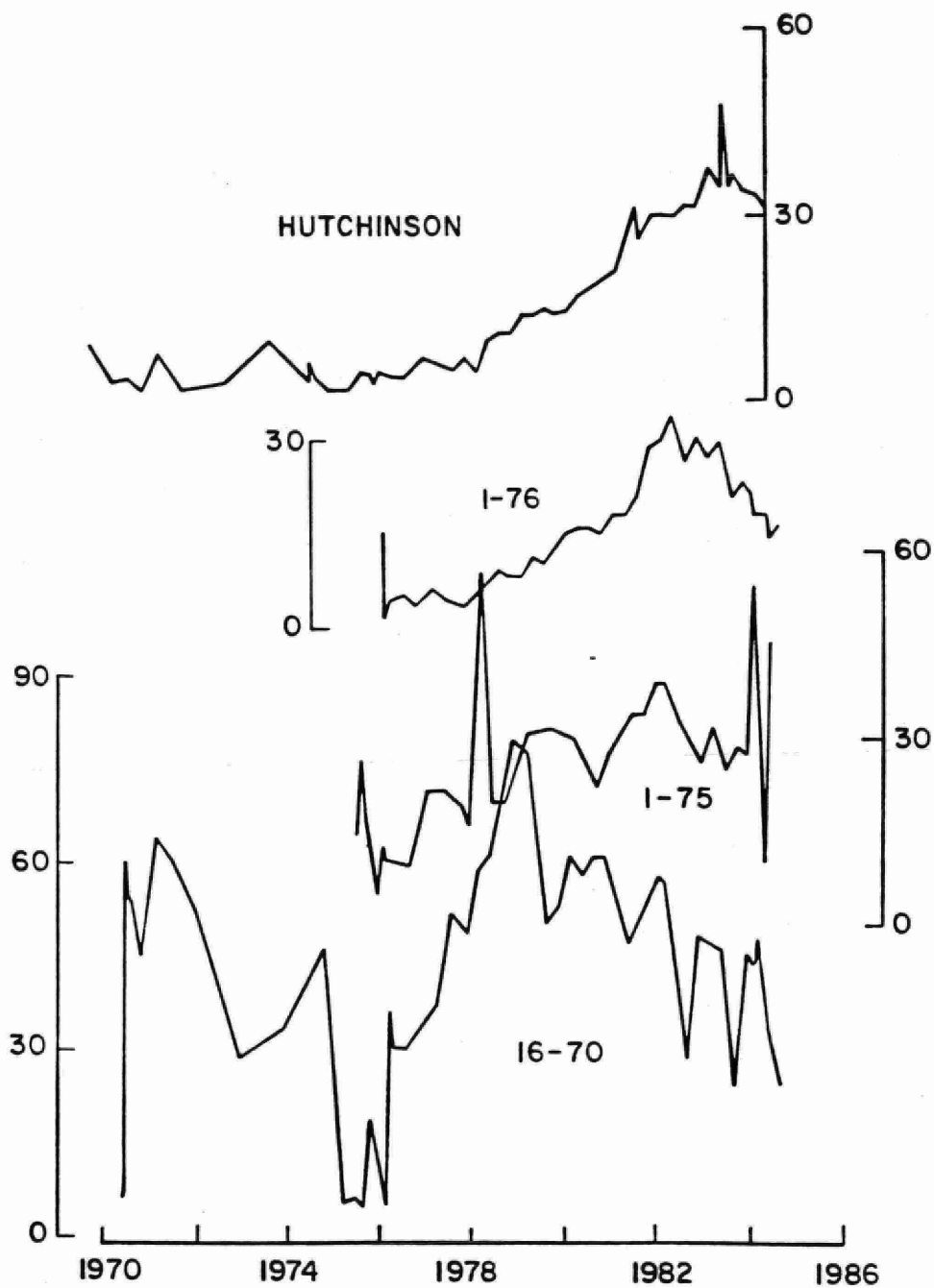
Figure 8:   Change in water quality due to contamination.   Chloride concentration (mg/l)
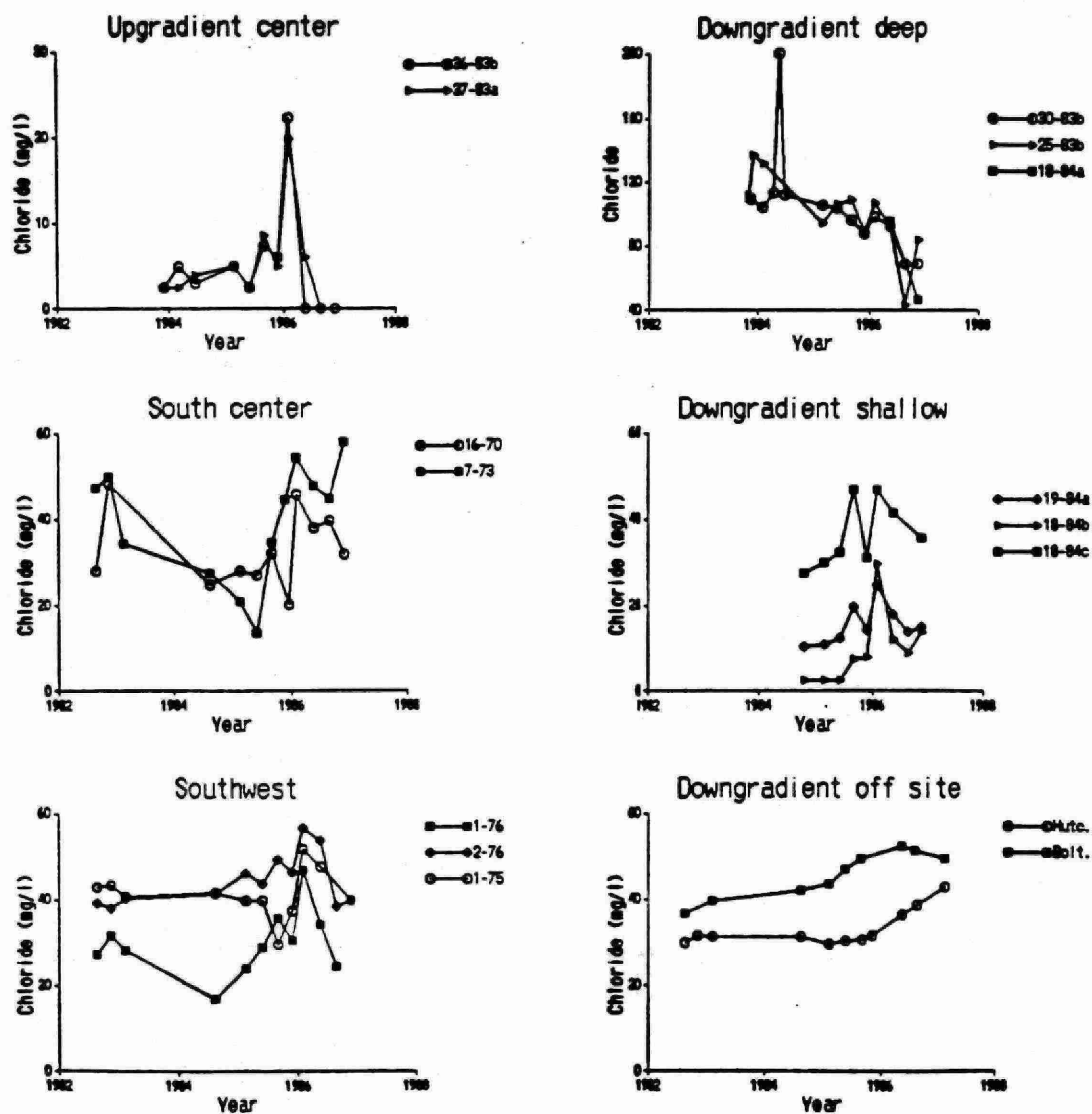
Figure 9:   Trends in chloride concentration
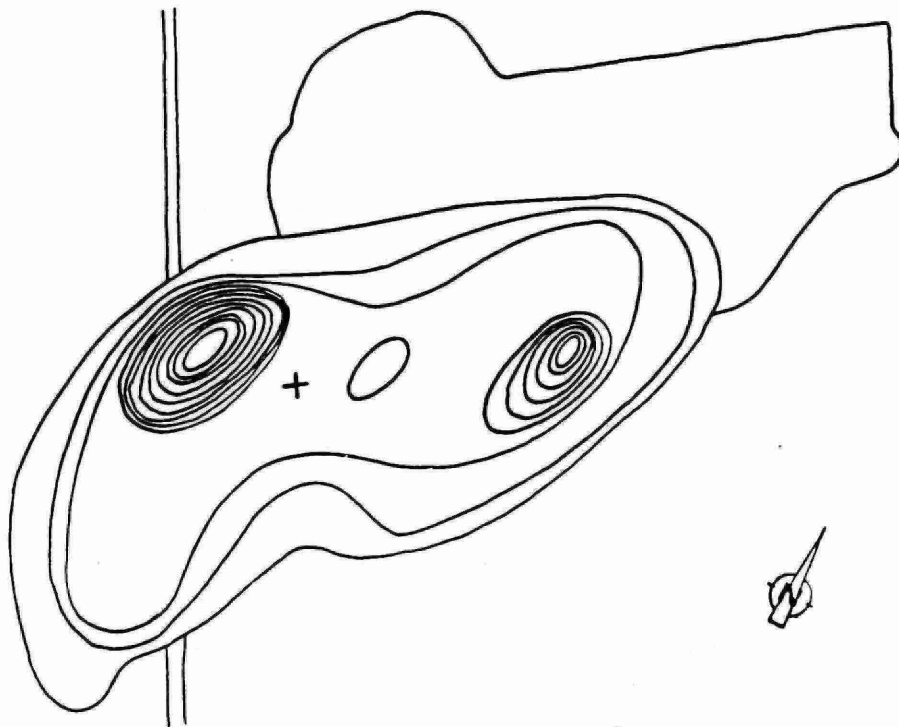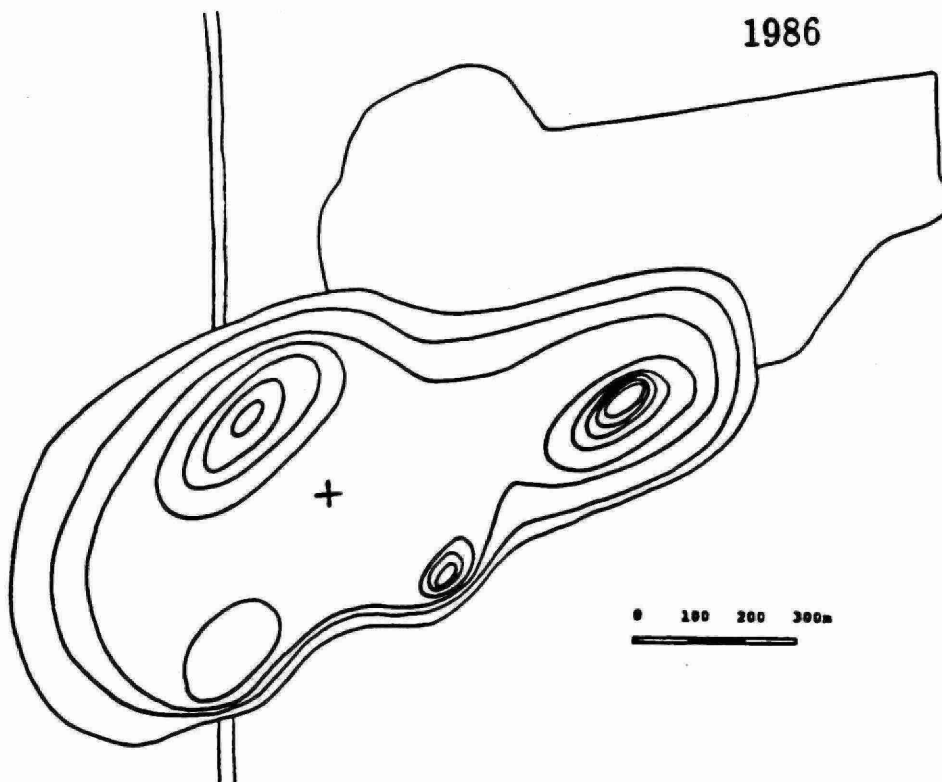
Figure 10:   Chloride plume for 1984 and 1986. Base contour of 20 mg/l and contour interval of 10 mg/l
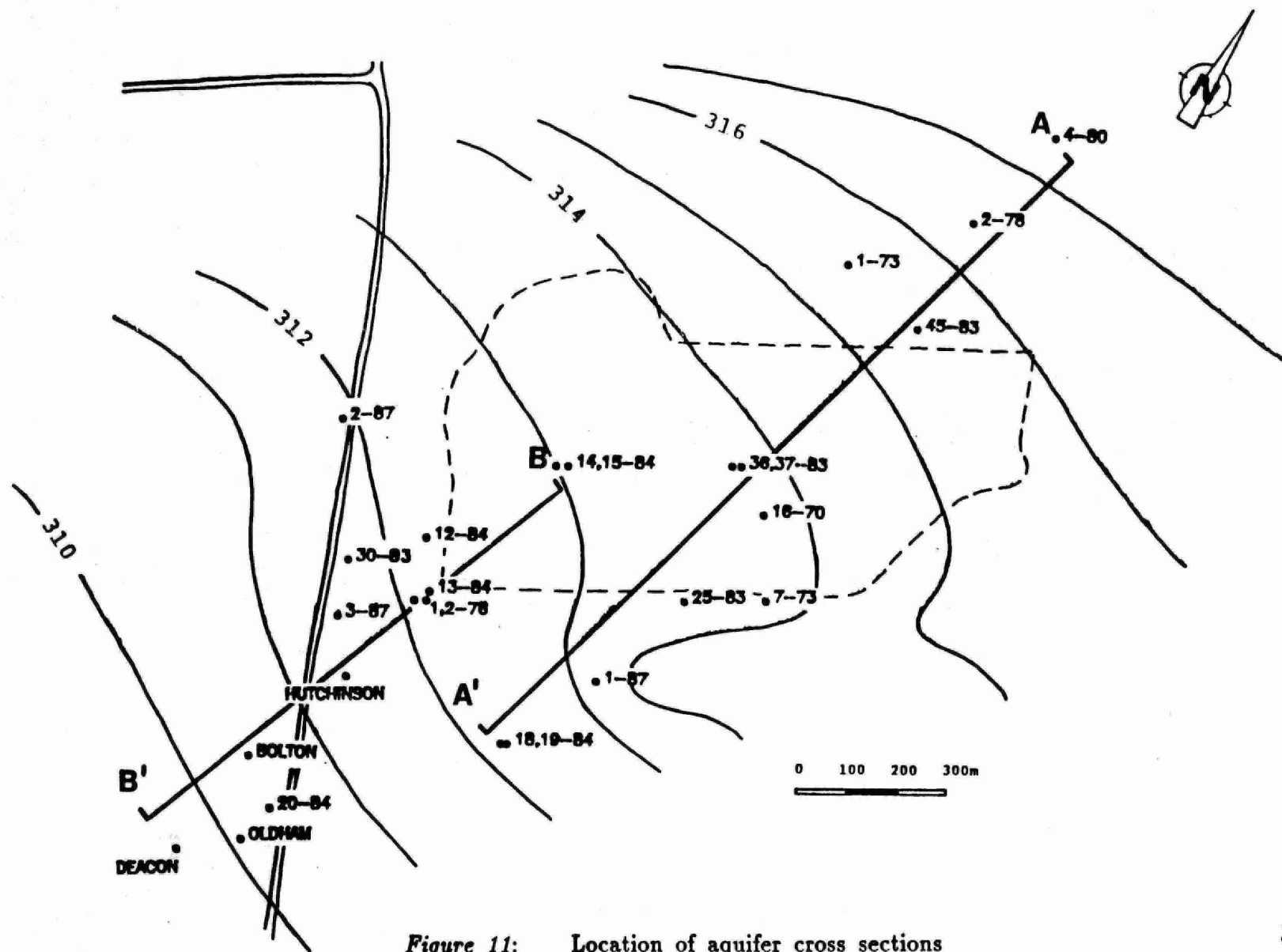
Figure 11: Location of aquifer cross sections

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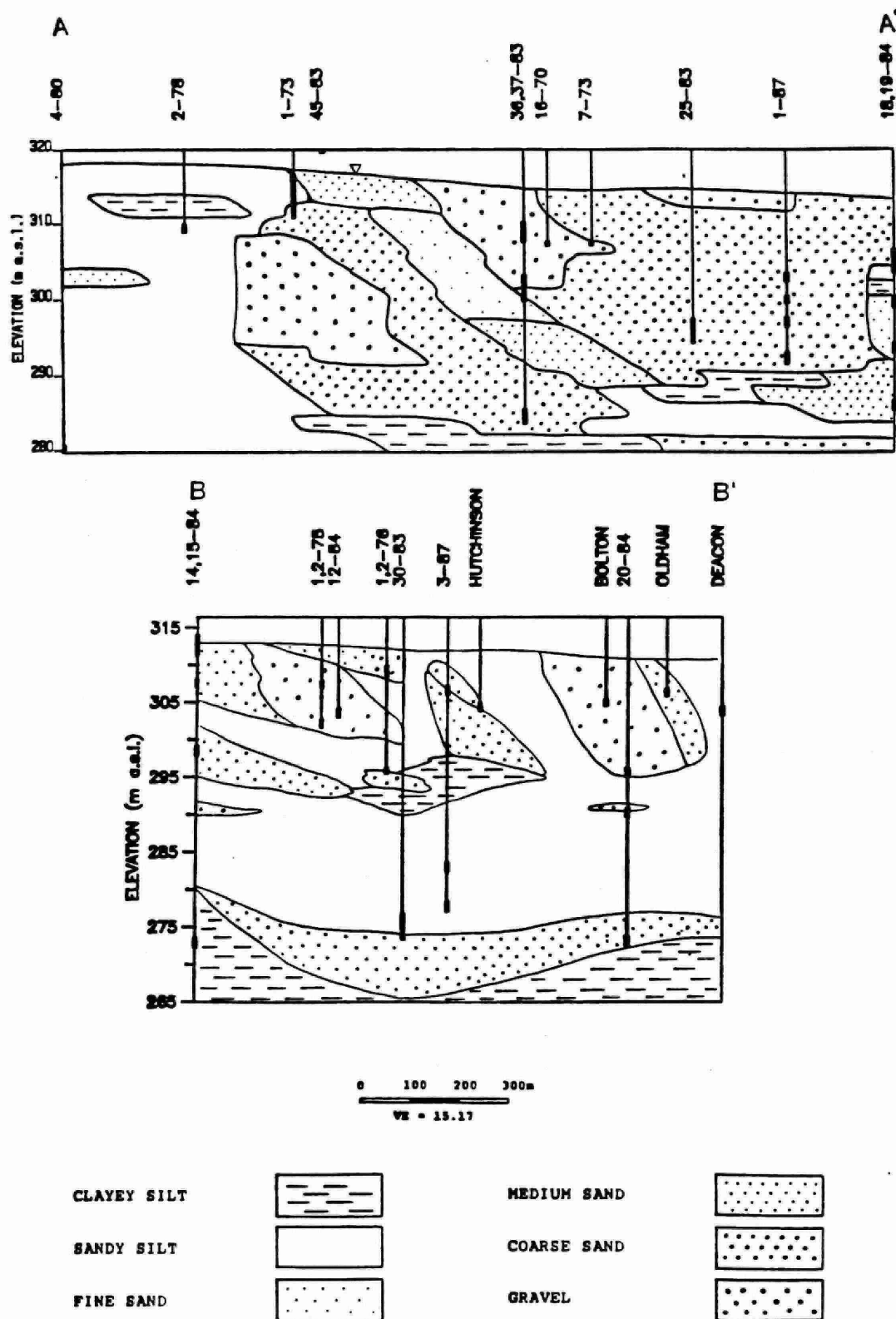
Figure 12:   Interpretation of aquifer stratigraphy
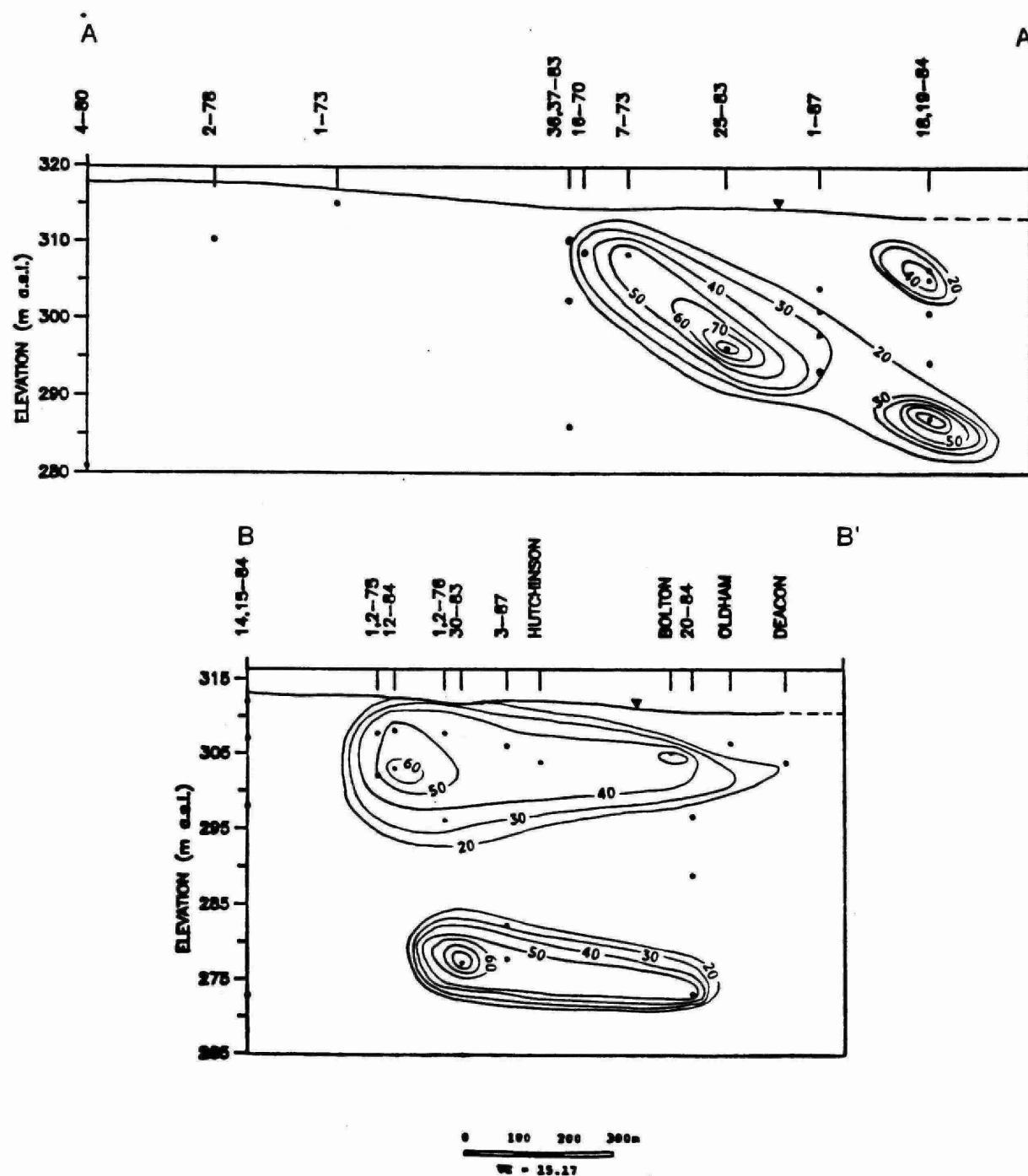
Figure 13:   Chloride plume in cross sections.
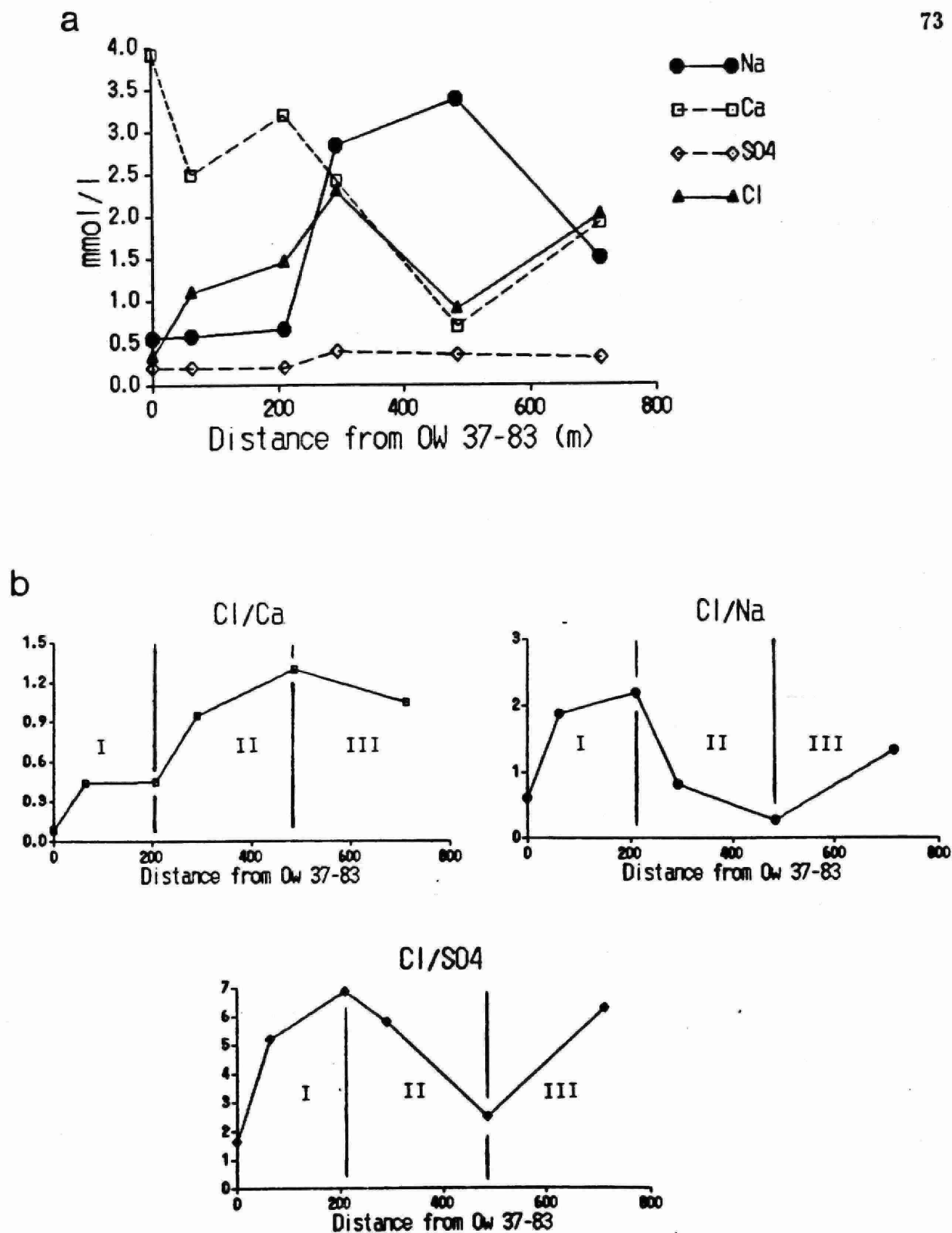
Figure 14: Variation in geochemistry along the flow path. a) Concentration of Na, Ca, SO4 and Cl in mmol/l b) Ratios Cl/species
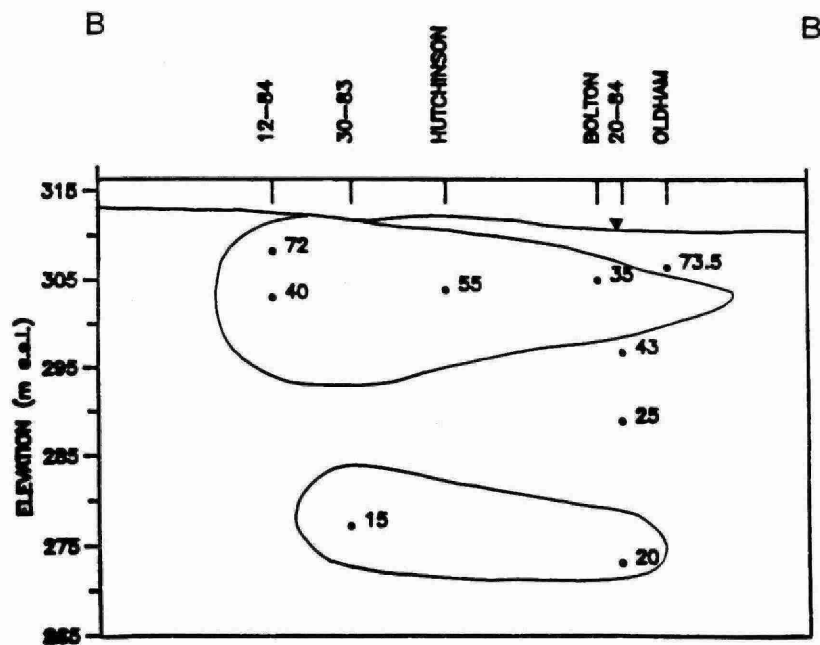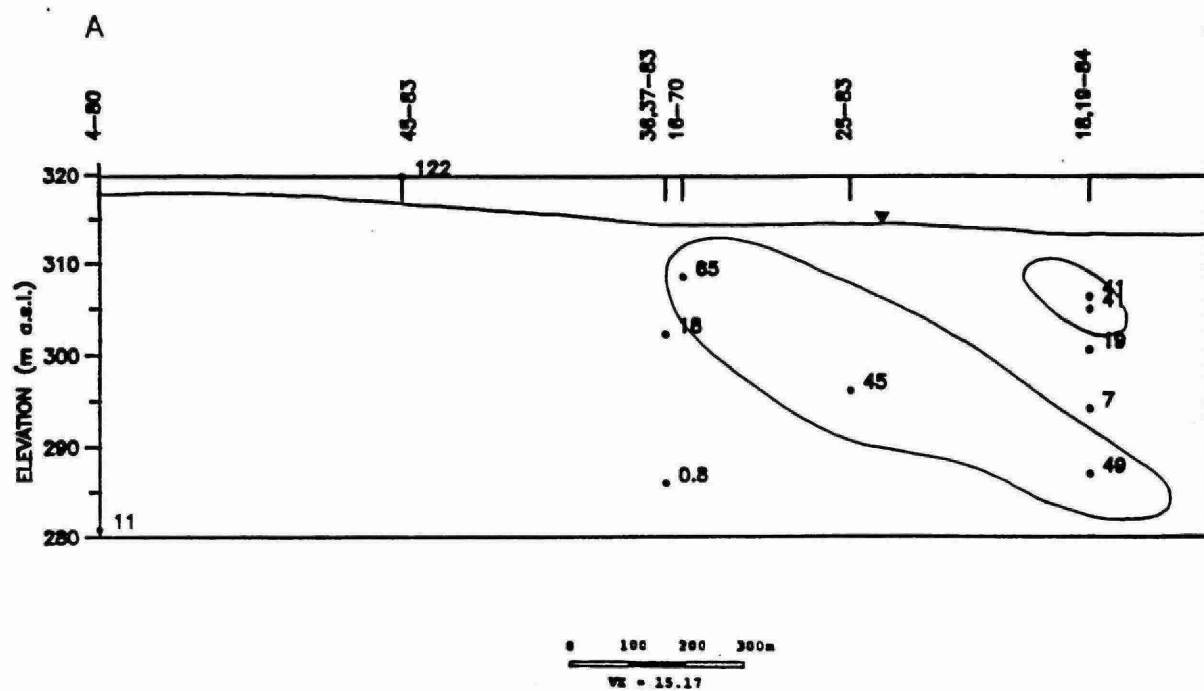
Figure 15: Tritium concentrations in aquifer cross sections. Interpreted chloride plumes are also shown.
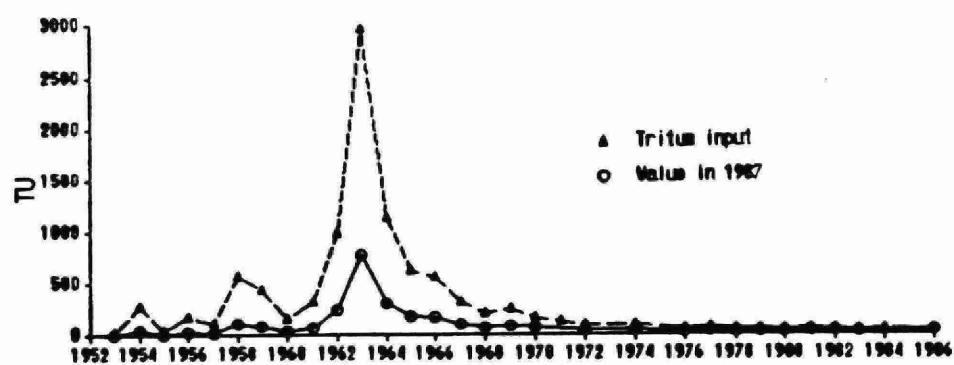
Figure 16:   Tritium concentration in precipitations
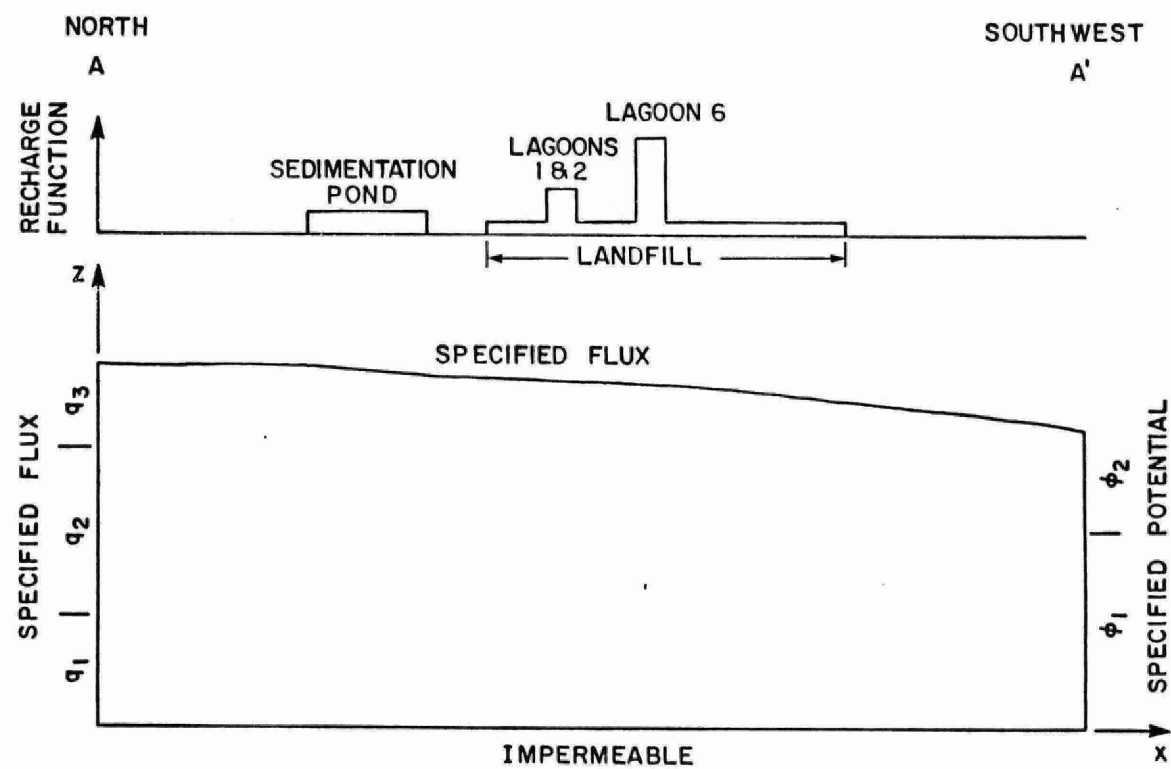
Figure 17:   Boundary conditions
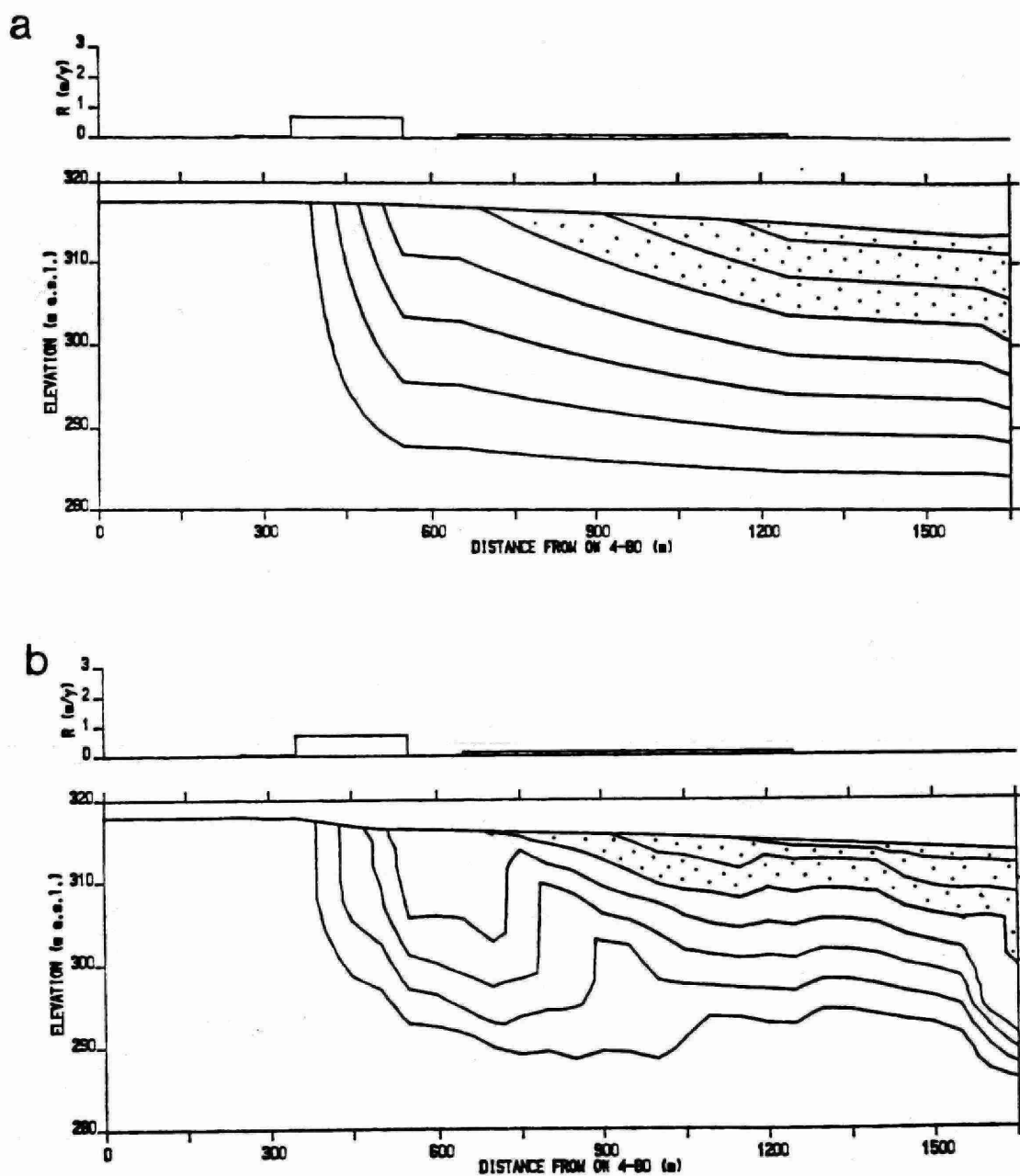
Figure 18: Flow patterns from calibration of the recharge function. a) Homogeneous stratigraphy b) Heterogeneous stratigraphy

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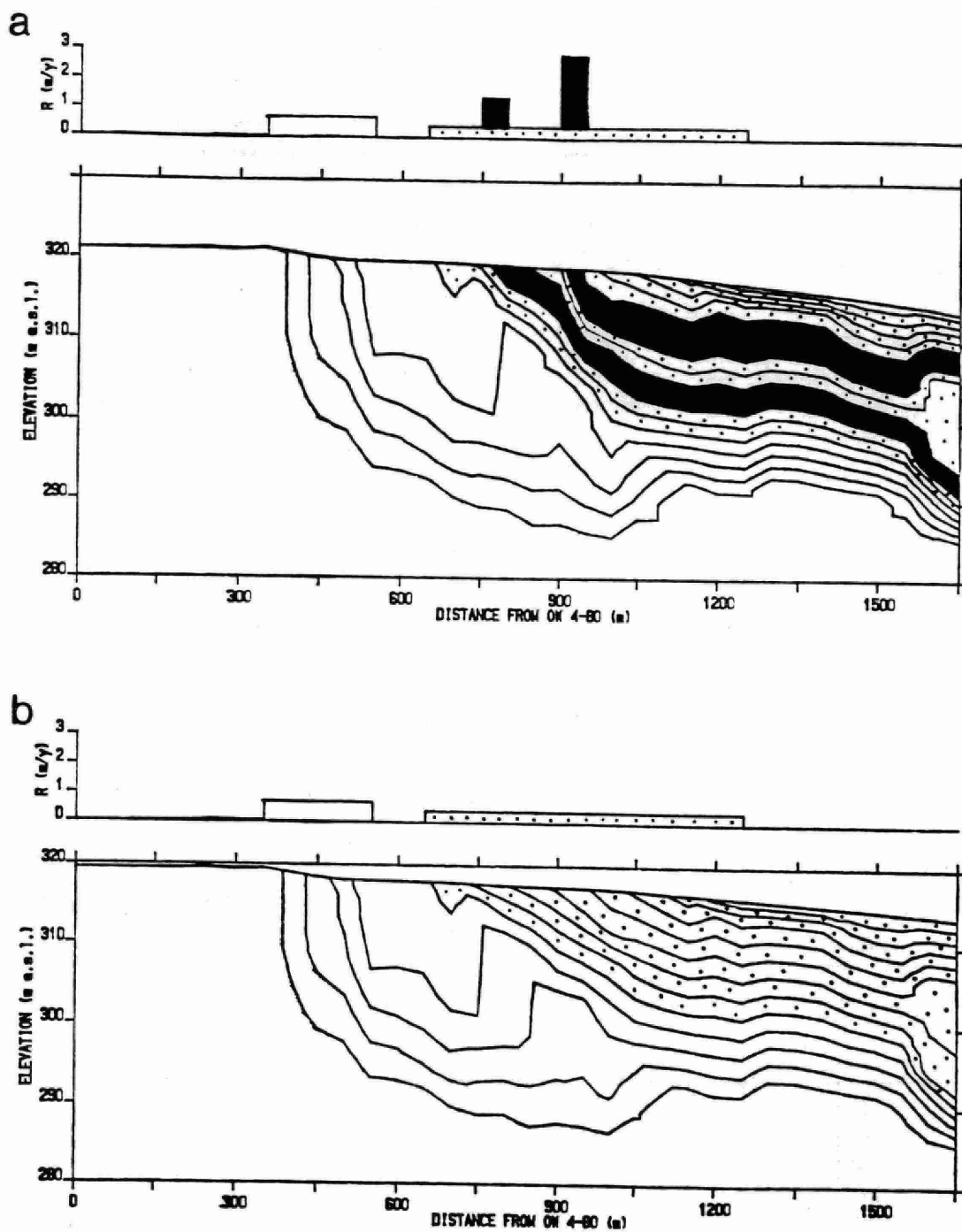
Figure 19: Flow patterns from simulation of operation conditions. a) Liquid and solid wastes (1962-1970) b) Solid wastes only (1970-1985)
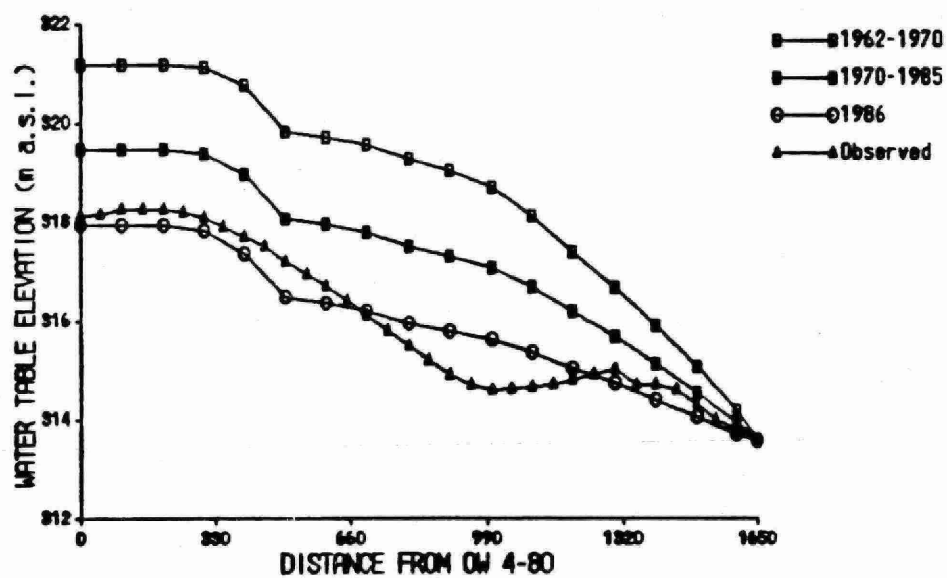
Figure 20:   Water table elevation from simulations
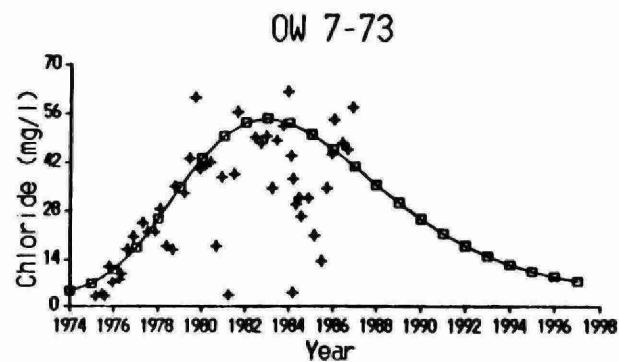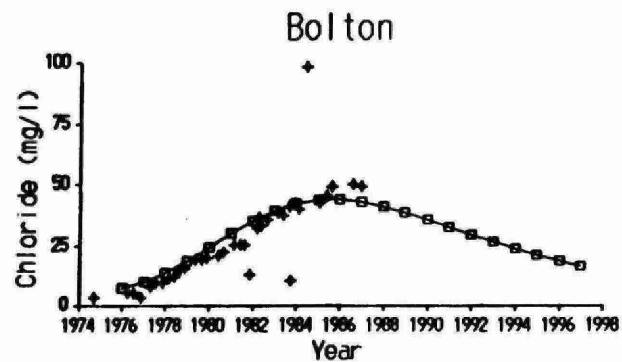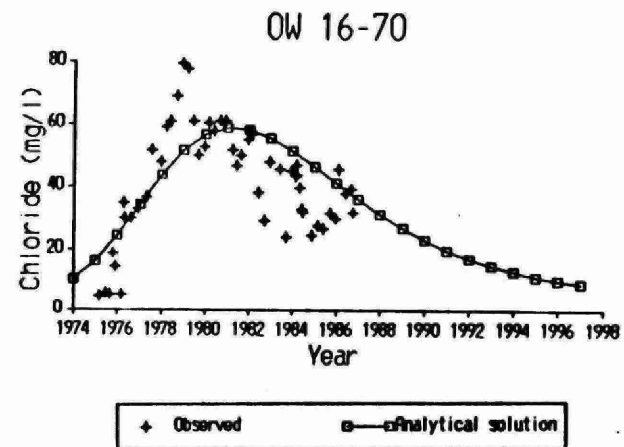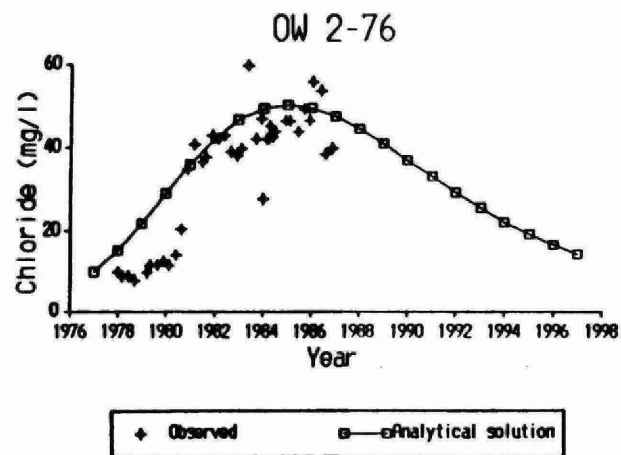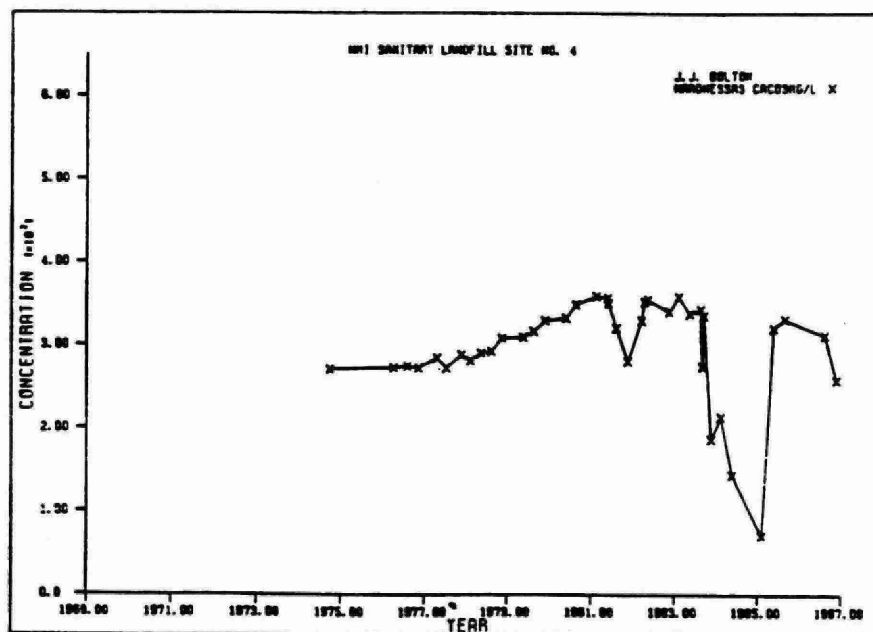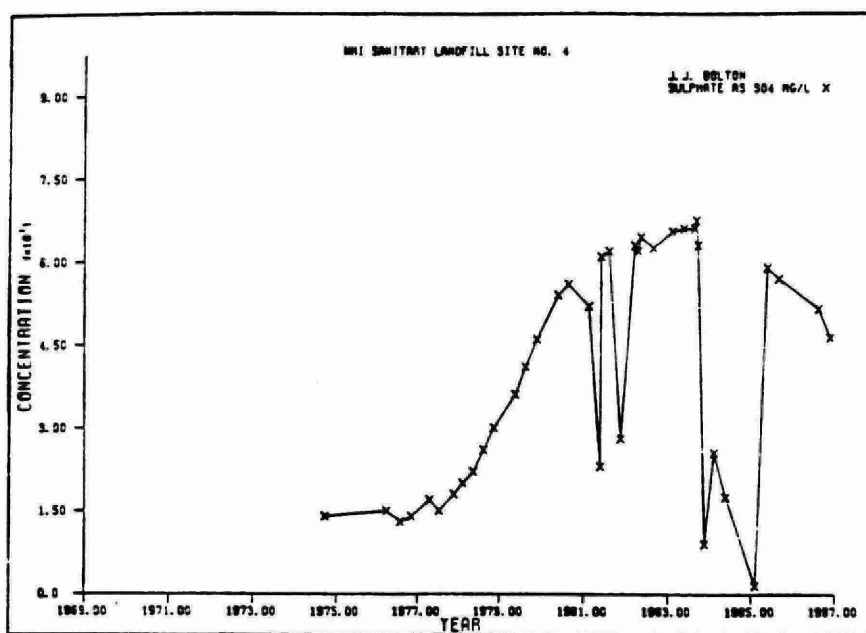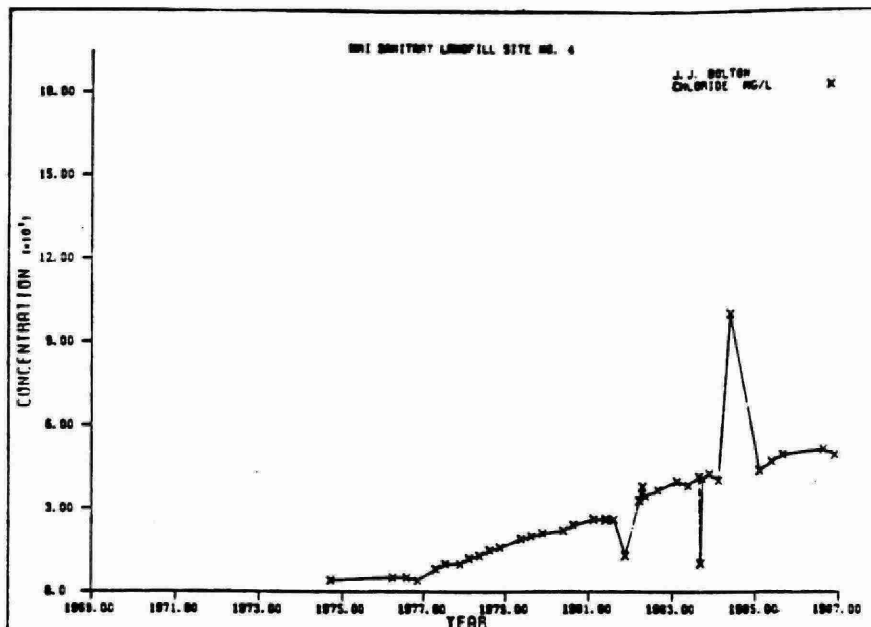
Figure 21: Curve fitting of chloride breakthrough curves

Appendix A
PHASE ONE DATA

(Figures from CRA (1985))

GWI SANITARY LANDFILL SITE NO. 4

BW16-70
CHLORIDE MG/L   X

GWI SANITARY LANDFILL SITE NO. 4

BW16-70
SULPHATE AS SO4 MG/L X

GWI SANITARY LANDFILL SITE NO. 4

BW16-70
HARDNESS AS CACO3MG/L X

82

84

WMI SANITARY LANDFILL SITE NO. 4

BH2-76
CHLORIDE MG/L   X

WMI SANITARY LANDFILL SITE NO. 4

BH2-76
SULPHATE AS SO4 MG/L X

WMI SANITARY LANDFILL SITE NO. 4

BH2-76
HARDNESS AS CACO3MG/L X

85

BRI SANITARY LANDFILL SITE NO. 4

J. J. BOLTON
CHLORIDE MG/L   X

BRI SANITARY LANDFILL SITE NO. 4

J. J. BOLTON
SULPHATE AS SO4 MG/L X

BRI SANITARY LANDFILL SITE NO. 4

J. J. BOLTON
HARDNESS AS CACO3 MG/L X

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WRI SANITARY LANDFILL SITE NO. 4

WRI SANITARY LANDFILL SITE NO. 4

WRI SANITARY LANDFILL SITE NO. 4

Appendix B
MAJOR ION CONCENTRATIONS UPGRADIENT

(Data used include analyses obtained by CRA (1985-87))

Northeast

West

Northeast

West

Northeast

Calcium (mg/l) vs Year

Legend: ◆–◆ 6-73, ○–○ 2-78, ■–■ 4-80

West

Calcium (mg/l) vs Year

Legend: ■–■ 6-78, ○–○ 6-82, ▶–▶ DEVLIN

Northeast

Magnesium (mg/l) vs Year

Legend: ◆–◆ 6-73, ○–○ 2-78, ■–■ 4-80

West

Magnesium (mg/l) vs Year

Legend: ■–■ 6-78, ○–○ 6-82, ▶–▶ DEVLIN

Northeast

West

Northeast

West

Northeast

West
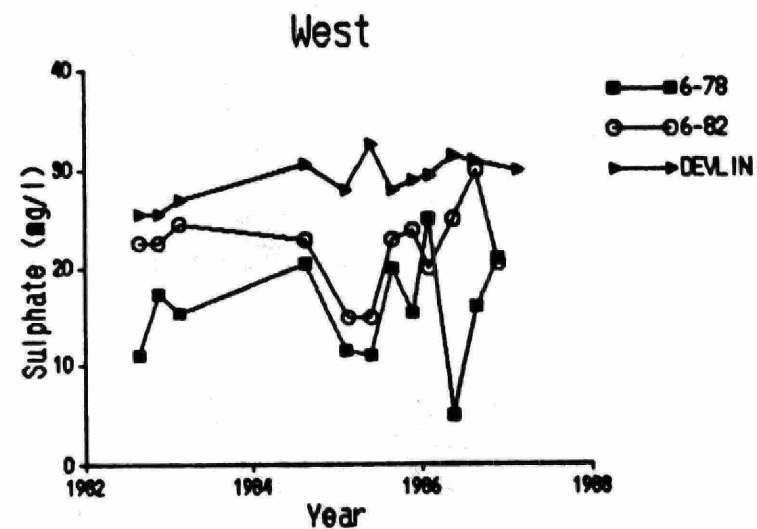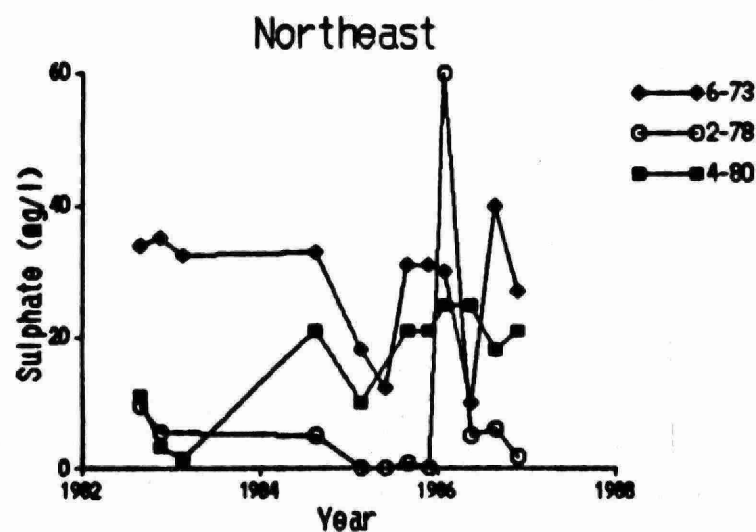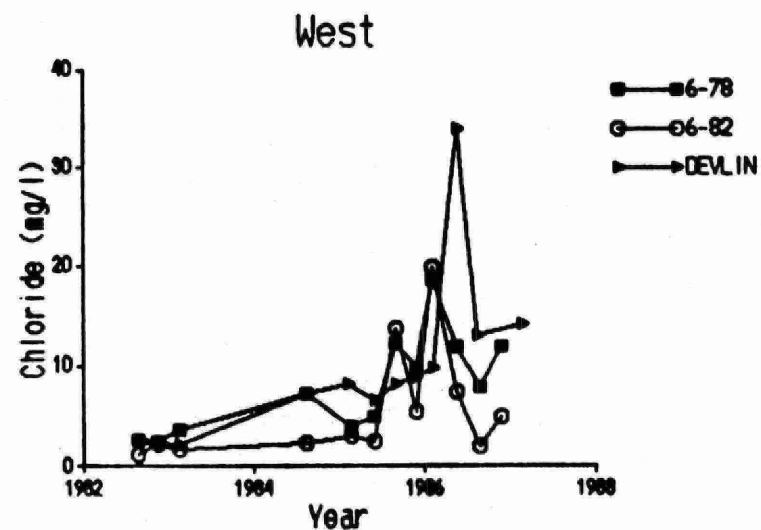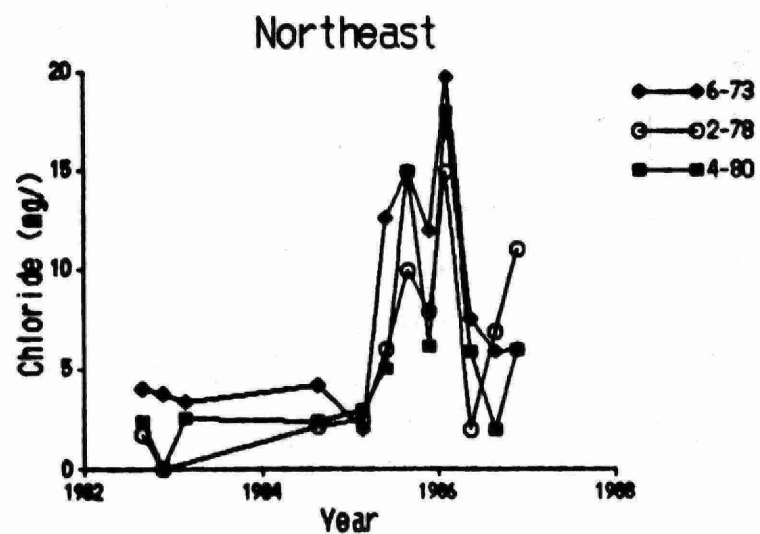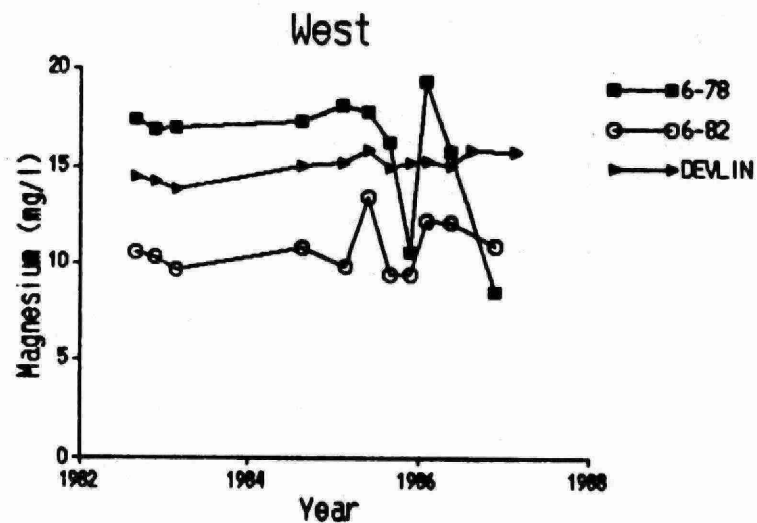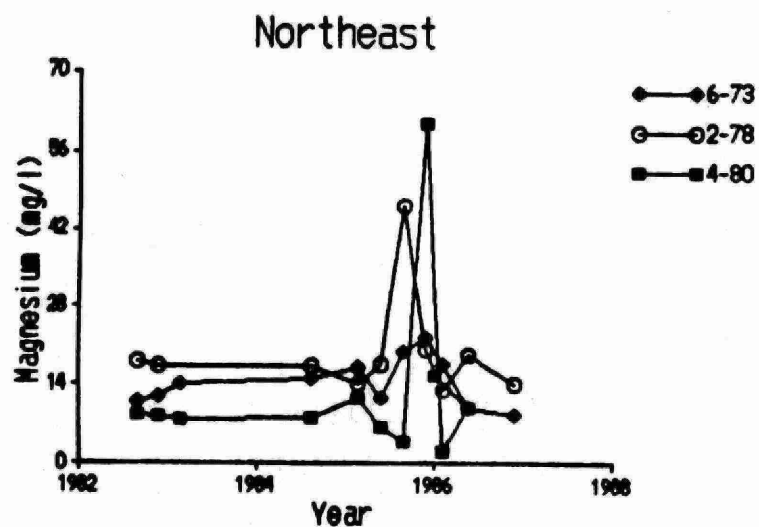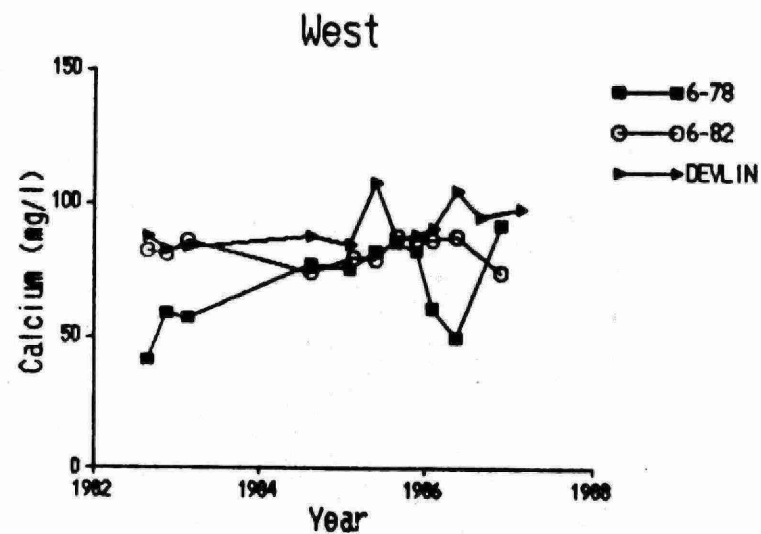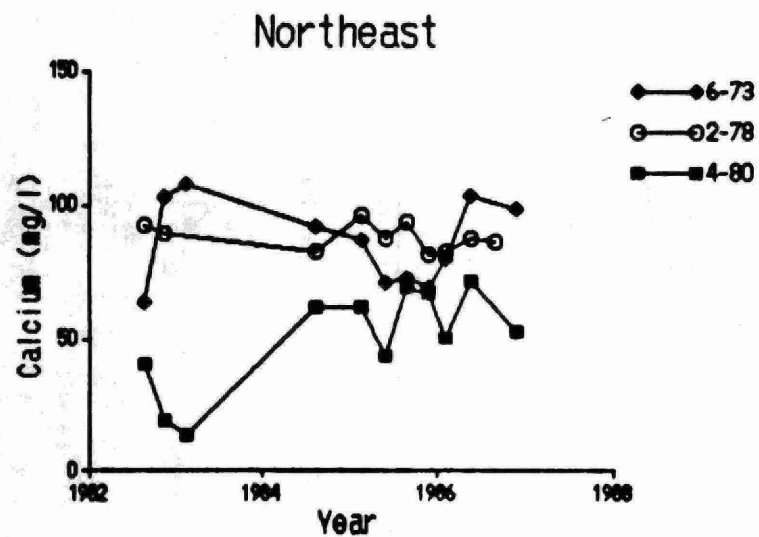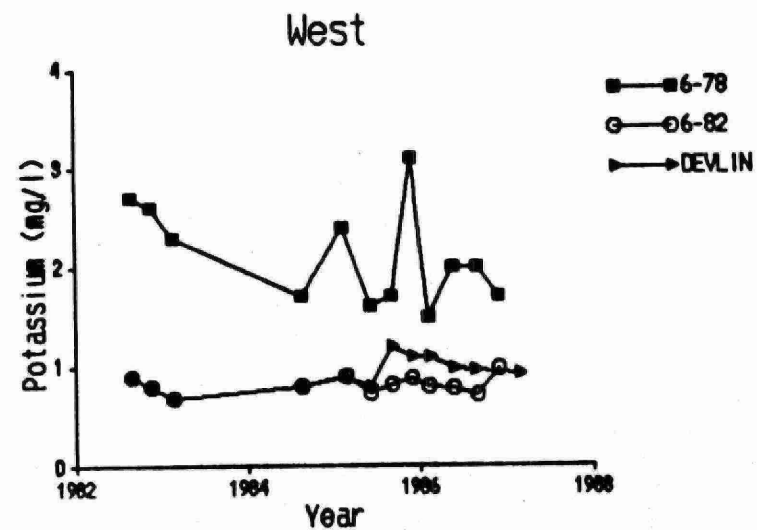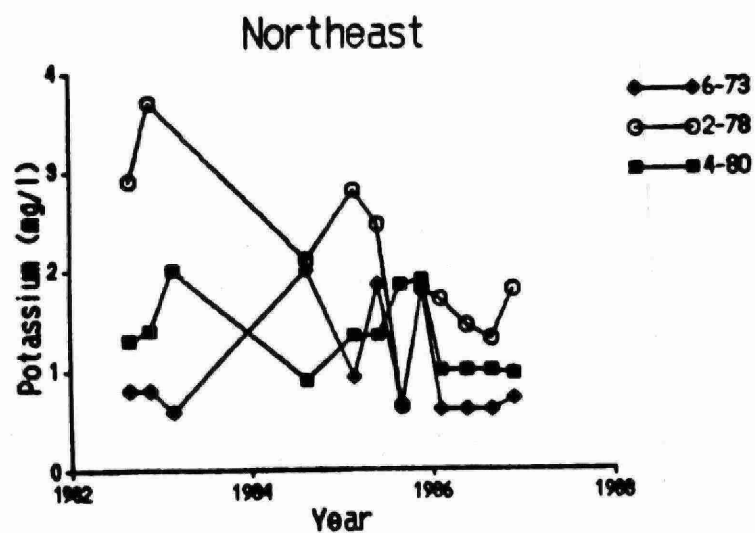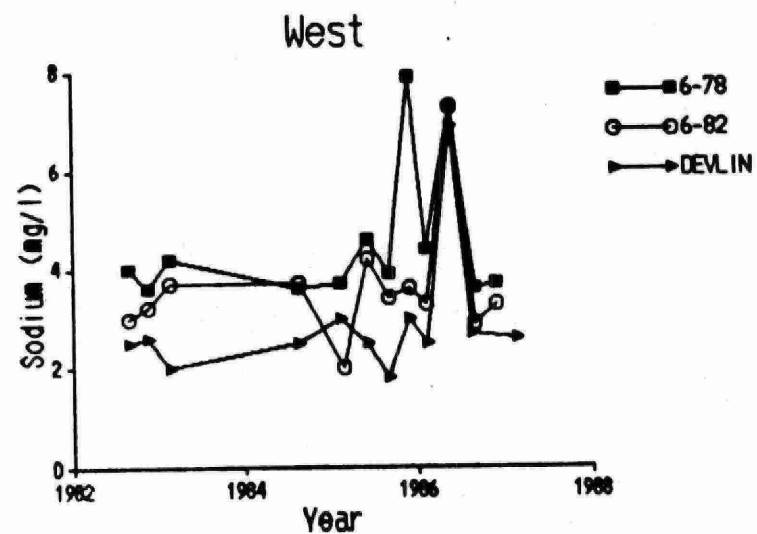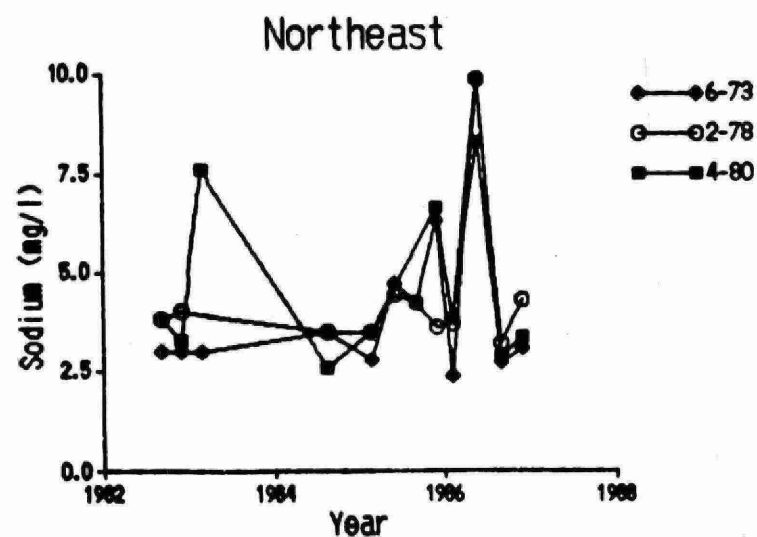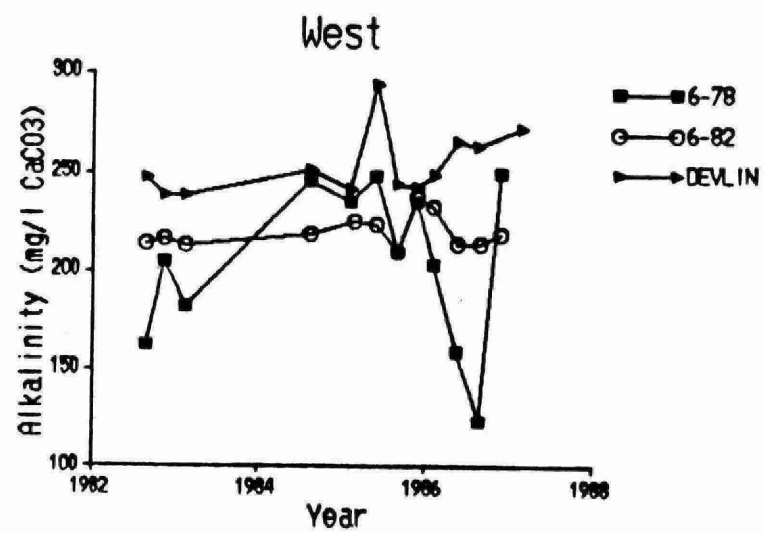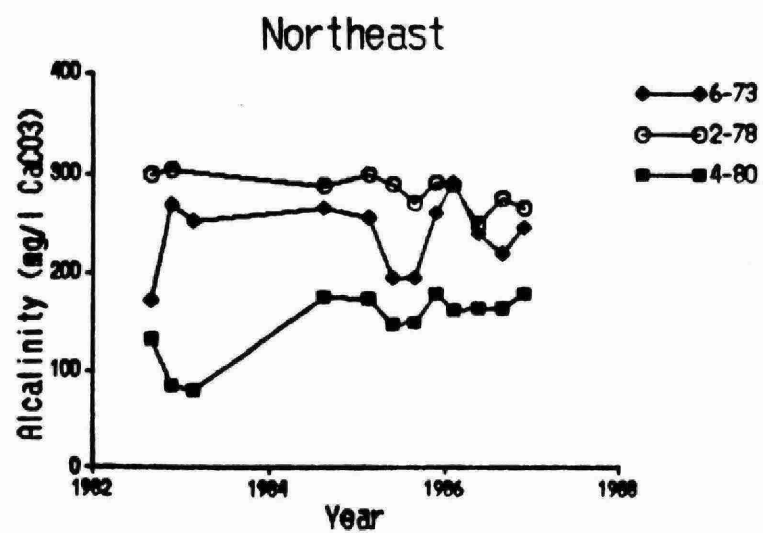
Appendix C
WATER QUALITY DATA AND CONTOURS

(Data used include analyses obtained by CRA (1985-87))

Figure 1: Aquifer observation wells

LEGEND

○ LYSI-78   LYSIMETER

□ SPI-78   SURFACE WATER SAMPLING LOCATION

● LWI-81   LEACHATE WELL MONITORING POINT

▲ SWI-83   SURFACE WATER ELEVATION MEASURING POINT

● BH24-83   BOREHOLE LOCATION

■ DP2a-83   DRIVE POINT

Figure 2:   Shallow observation wells and boreholes

Table 1: Values used in contour maps for 1982

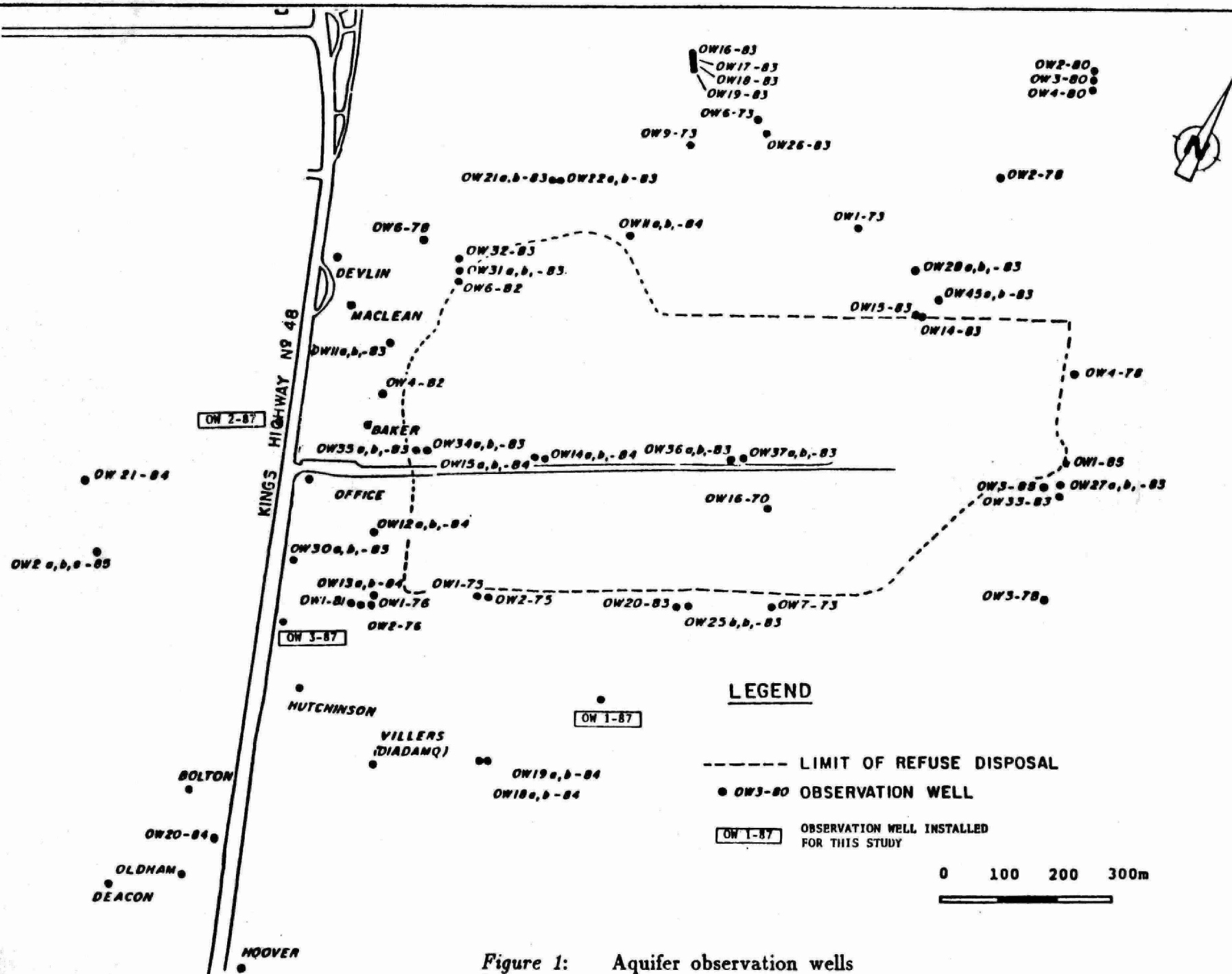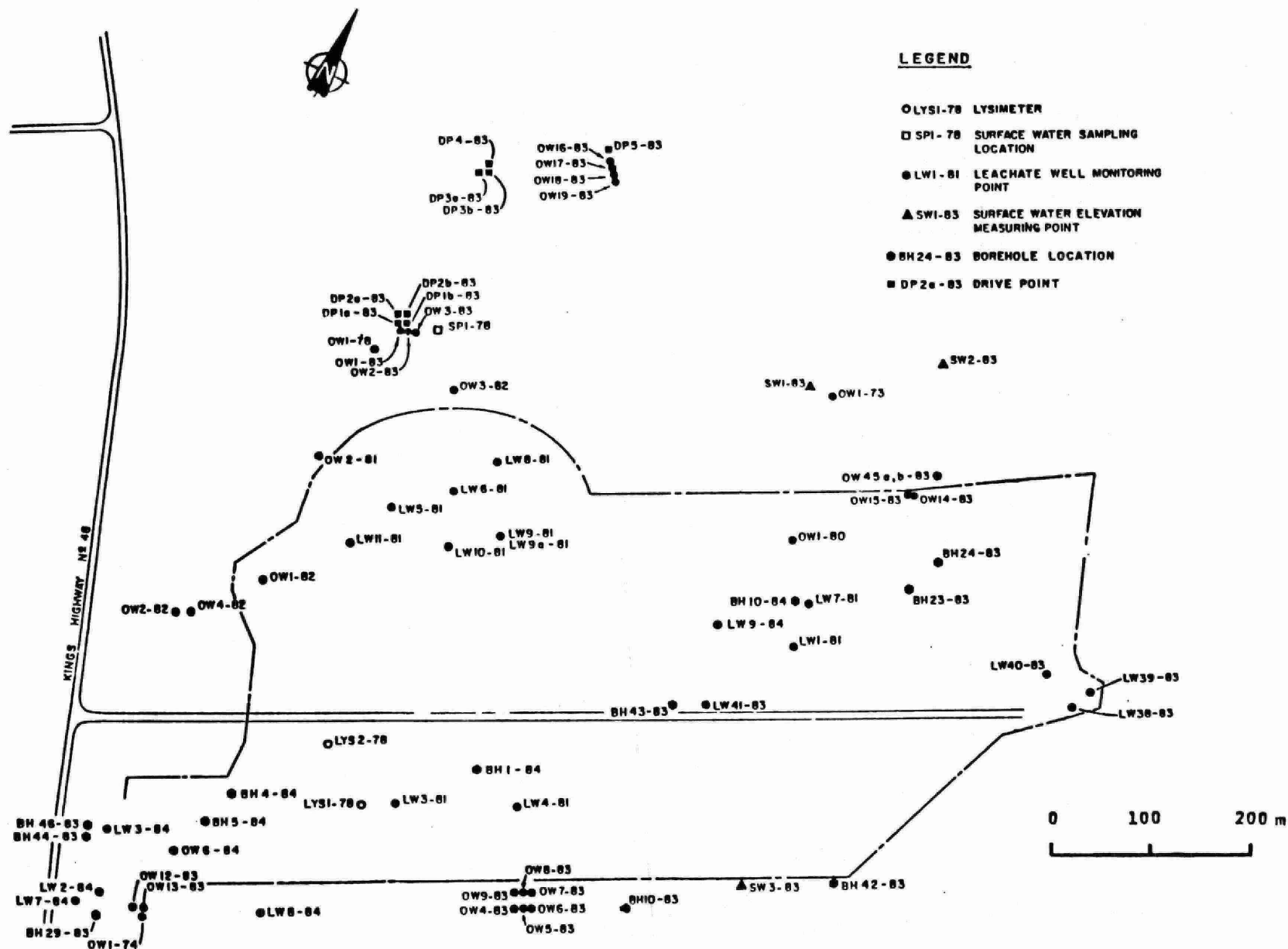| Well	Ca	Mg	Na	K	Alk.	SO4	Cl
16-70	106.0	16.0	41.0	2.90	307	57.0	42.0
1-73	109.5	9.4	22.8	1.55	280	42.0	22.0
6-73	83.5	11.5	3.0	0.70	220	34.5	5.0
7-73	124.5	12.0	46.8	1.70	338	37.0	51.0
9-73	94.5	14.9	3.3	0.85	261	37.5	6.0
1-75	41.0	12.1	19.0	1.15		6.0	30.0
2-75	73.3	20.1	24.0	2.15	243	7.0	45.0
1-76	55.0	16.6	7.7	2.30		23.0	30.0
2-76	120.0	25.3	15.0	3.40	347	30.0	42.0
2-78	91.0	17.7	3.9	3.30	302	7.5	4.0
3-78	72.0	11.1	3.4	1.05		19.0	2.0
6-78	50.0	17.2	3.8	2.65		14.2	3.0
2-80	60.0	16.3	5.5	8.60		33.0	4.0
3-80	35.9	11.0	4.6			7.0	3.0
4-80	30.0	8.6	3.6	1.35	110	7.0	2.0
1-81	7.0	5.7	34.3	6.60	77	30.0	17.0
4-82	58.0	11.5	9.0	3.40	231	25.5	2.0
6-82		10.4		0.85	215		
Baker	92.3	12.4	3.5	0.80	255	27.5	6.0
Bolton	88.5	17.8	15.8	1.00	212	62.5	28.0
Deacon					230		13.0
Devlin	85.0	14.3	2.5	0.85	243		2.0
Hoover	85.0	14.3	3.5	0.90	242	21.0	3.0
Hutchinson	111.5	20.0	9.3	1.25	292	57.0	33.0
Oldham	81.0	13.6	3.6	0.90	229	21.5	6.0
Villers	87.3	14.7	3.6	1.10	255	21.0	6.0

Note: all values in mg/l, alkalinity as CaCO3.

Table 2: Values used in contour maps for 1984

Well	Ca	Mg	Na	K	Alk.	SO4	Cl
16-70	83.0	13.0	46.5	3.10	309	29.0	37.7
1-73	131.0	10.2	15.5	1.80	325	50.0	17.8
6-73	92.0	15.0	3.5	2.00	265	33.0	4.2
7-73	107.0	14.7	28.2	1.80	342	16.0	33.1
9-73	74.5	12.8	2.5	1.20	194	35.0	10.0
2-75	94.0	18.0	23.0	1.70	274	32.5	46.0
1-76	74.0	13.4	12.5	1.40	219	32.0	19.9
2-76	114.0	23.5	20.5	2.50	363	19.0	40.2
2-78	83.0	17.3	3.5	2.10	288	5.0	2.2
6-78	77.5	17.3	3.6	1.70	246	20.5	6.7
4-80	62.5	8.2	2.6	0.90	176	21.0	2.4
1-81	83.0	13.8	10.0	1.10	231	35.0	23.0
4-82	91.5	12.0	4.0	1.40	254	29.0	4.0
6-82	74.5	10.8	3.7	0.80	219	23.0	2.4
20-83					310	42.0	51.5
25-83b					314	56.0	95.0
30-83b					480	< 5.	126.0
34-83b					155	19.5	14.2
35-83b					322	20.0	12.0
37-83b					367	26.4	9.3
11-84b					61	17.5	
12-84a					554		84.5
12-84b						47.0	55.0
13-84a							2.0
13-84b					287	34.0	32.0
18-84c					270	9.0	27.5
19-84b					283	5.0	27.0
20-84a					213	40.0	25.0
21-84b						35.0	
21-84c					252		9.5
Baker	96.0	12.1	2.9	0.90	266	24.0	6.0
Bolton	96.0	16.2	32.8	1.30	265	57.0	4.0
Deacon	79.0	15.4	4.1	1.10	238	18.0	16.0
Devlin	87.5	15.1	2.5	0.80	252	30.5	7.0
Hoover	78.0	14.8	2.9	1.00	248	16.0	5.0
Hutchinson	125.0	20.5	16.2	1.40	351	59.0	34.0
Oldham	87.0	13.8	3.5	1.00	208	18.0	8.0
Villers							15.0

Note: all values in mg/l, alkalinity as CaCO3.

Table 3: Values used in contour maps for 1986

Well	Ca	Mg	Na	K	Alk.	SO4	Cl	DOC
16-70	99.6	9.8	13.4	2.20	256	20.3	38.9	3.1
1-73	107.0	12.8	19.5	2.40	293	41.0	33.8	8.0
6-73	82.0	12.9	4.5	0.64	249	26.8	9.8	1.8
7-73	128.2	20.7	15.1	1.20	350	20.5	51.5	3.1
2-75	101.7	16.0	22.5	1.30	274	24.3	46.6	5.0
1-76	79.2	15.3	14.6	1.30	237	26.3	32.3	2.9
2-76	104.3	22.5	23.3	1.65	345	12.8	47.4	3.0
2-78	84.7	16.0	5.3	1.60	269	17.8	8.8	5.0
6-78	60.9	12.8	4.8	1.80	184	16.8	12.6	16.0
4-80	57.5	8.7	4.6	1.00	167	22.5	8.1	1.4
4-82	93.2	13.4	5.6	1.20	261	21.5	11.5	2.3
6-82	82.1	11.0	4.2	0.82	220	23.9	8.7	1.9
25-83b	97.3	17.5	65.3	2.70	329	38.8	81.8	4.5
27-83a	76.3	10.2	4.6	1.20	234	0.0	13.7	2.5
30-83b	65.5	16.7	138.8	2.60	436	26.8	82.1	18.5
31-83b	80.5	9.0	4.6	1.20		25.2	4.1	1.6
33-83	86.8	12.4	4.2	1.10	275	14.4	9.8	
35-83b	135.3	27.2	11.5	5.00	402	21.3	28.7	1.8
36-83b	46.7	12.1	19.5	1.90	172	1.6	7.1	7.8
37-83b	157.0	12.7	12.9	1.20	177	20.0	9.4	2.4
11-84b	48.3	10.5	5.1	2.50	100	26.0	38.5	3.5
12-84a	111.0	24.3	97.8	2.30	507	31.0	63.7	12.1
18-84a	76.5		34.7	2.45	172	30.0	71.4	2.3
18-84b	17.0		27.3		56	46.8	16.1	4.0
18-84c	95.7	15.0	22.3	1.30	297	5.0	41.7	1.6
19-84a	66.5	13.4	5.7	1.50	219	1.5	17.9	3.0
19-84b	92.9	14.4	20.7	1.70	298	0.5	39.6	1.9
20-84a	106.0	12.6	14.4	1.60	284	84.0	51.2	3.8
20-84b	50.6	6.8	5.8		148	10.0	7.0	1.8
20-84c	77.4		5.4		221	15.4	16.7	1.9
21-84a	69.1	4.8	4.5	1.00	189	13.5	11.6	2.4
21-84b	45.9	14.9	6.4	2.60	148	28.5	5.0	1.5
21-84c					239			
2-85a	25.0	10.3	8.3	7.00	118	10.5	7.7	3.0
2-85b	55.8	8.7	8.9	4.30	174	15.8	12.4	2.5
2-85c	107.0	14.6	14.6	2.20	305	39.8	23.8	5.4
3-85	75.0	10.2	3.4	0.83		0.0	7.9	
1-87a	16.4	2.0	58.3	0.90	237	24.5	32.0	8.5
1-87b	27.9	5.0	77.8	1.50	411	35.0	32.3	11.0
1-87c	18.1	3.0	67.4	0.80	291	35.9	22.9	8.5
1-87d	14.6	2.0	58.0	0.70	291	29.8	17.2	9.0
2-87a	40.0	5.0	37.0	2.00	234	9.2	59.5	17.0
2-87b	49.0	9.0	59.0	2.00	293	22.6	88.2	13.0
2-87c	67.0	14.0	80.0	2.60	351	33.9	87.5	13.0

3-87a	54.0	6.0	39.0	2.60	314	10.1	50.6	11.0
3-87b	51.0	6.7	29.0	2.30	355	6.6	30.0	10.0
3-87c	50.0	6.7	48.0	1.90	393	16.6	42.2	20.0
Baker	95.0	13.8	3.9	0.95	251	23.5	14.6	
Bolton	87.9	15.8	37.6	1.20	260	49.0	50.7	1.9
Deacon	96.9	15.6	4.2	1.00	250	23.4	19.6	0.8
Devlin	97.4	15.5	3.7	1.00	263	30.5	17.8	1.4
Hoover	54.8	13.5	3.3	1.00	185	7.4	8.3	0.4
Hutchinson	129.8	20.8	17.0	1.40	325	46.9	40.9	1.1
Oldham	76.9	12.5	3.7	1.10	221	19.7	5.8	0.6

Note: all values in mg/l, alkalinity as CaCO3.

Table 4: Upgradient water quality

1- Mean

Year	Cl	SO4	Ca	Mg	K	Na	Alk.	DOC
1982	3.0	18.5	70.0	13.3	1.64	3.3	209	
1984	4.6	22.2	79.6	14.0	1.40	3.2	241	
1986	11.0	23.0	79.5	12.7	1.14	4.5	225	4.6

2-Standard deviation

Year	Cl	SO4	Ca	Mg	K	Na	Alk.	DOC
1982	0.9	10.9	24.4	3.7	1.09	0.5	66	
1984	2.0	9.9	10.5	3.7	0.60	0.5	39	
1986	3.7	5.2	16.8	2.8	0.42	0.5	43	5.8

Note: arithmetic mean and standard deviation for observation
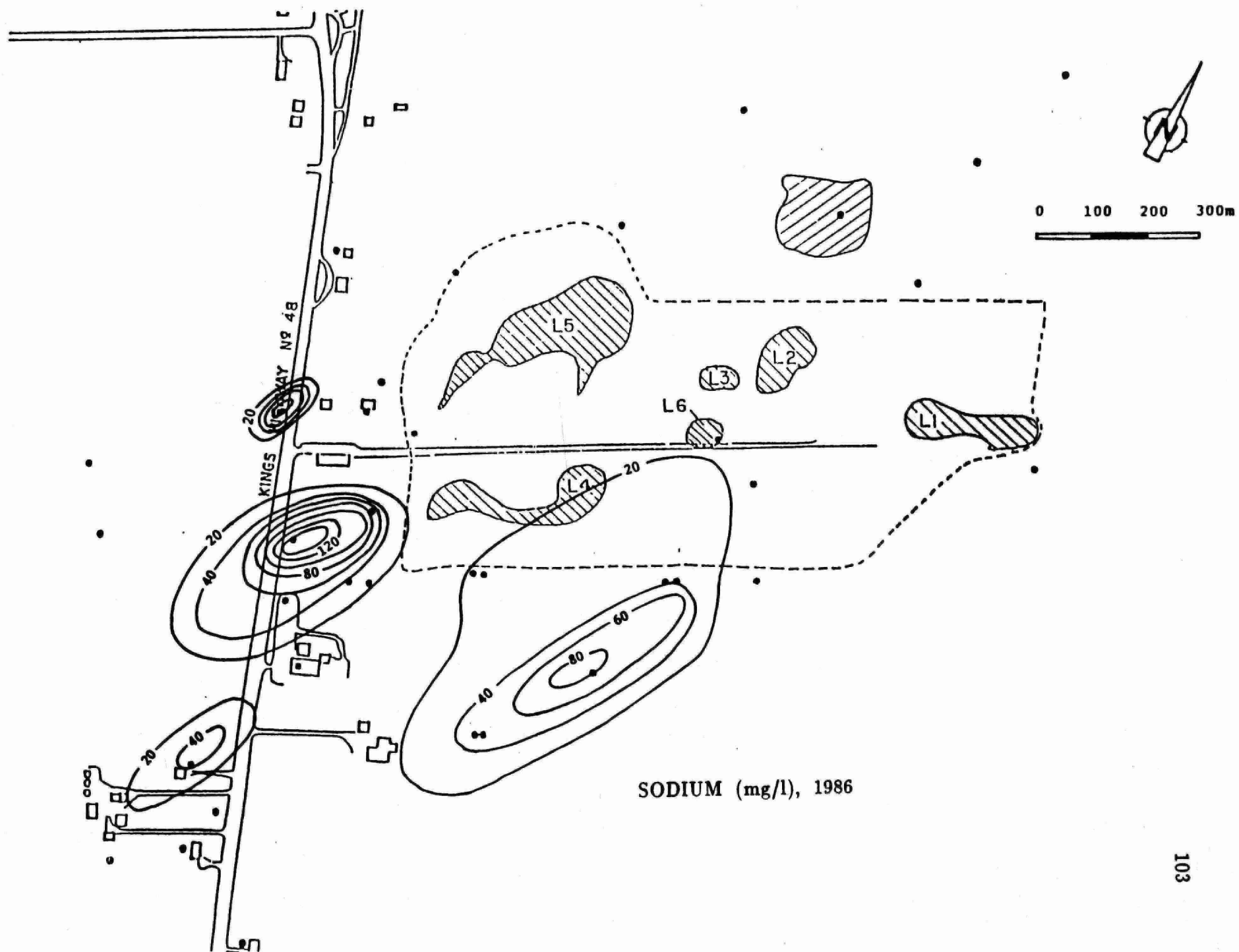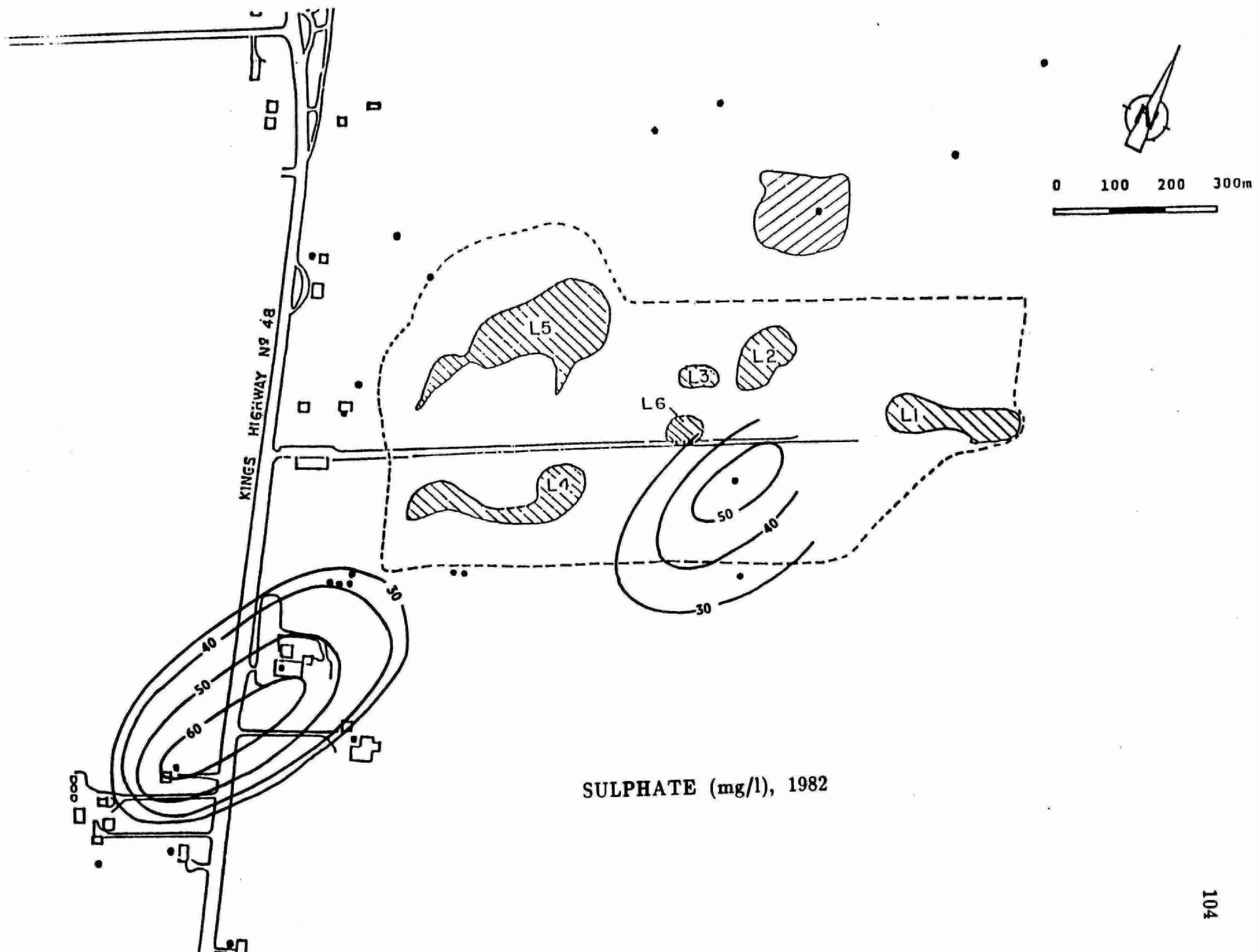wells 6-73, 2-78, 4-80, 6-78, 6-82, Devlin
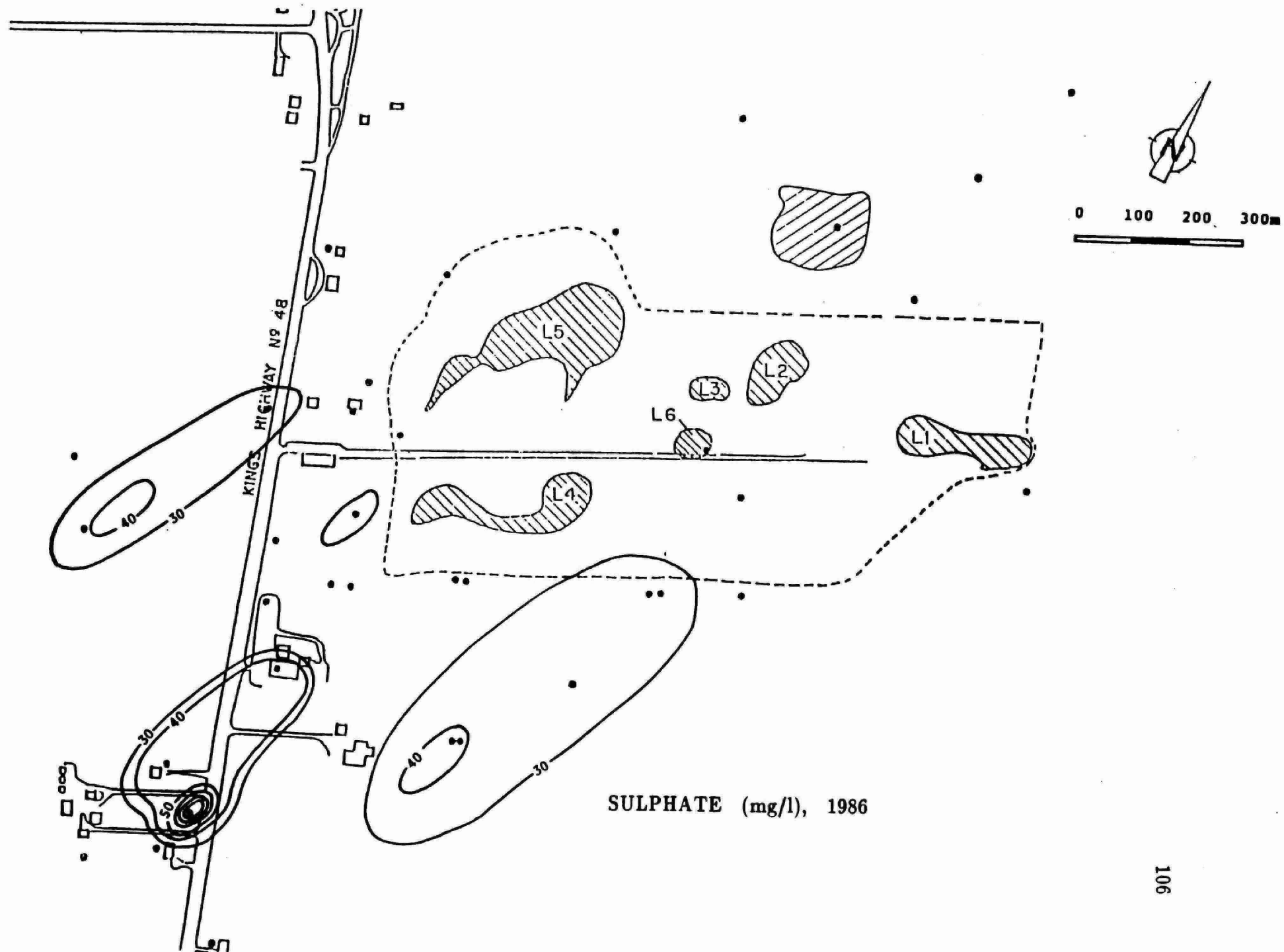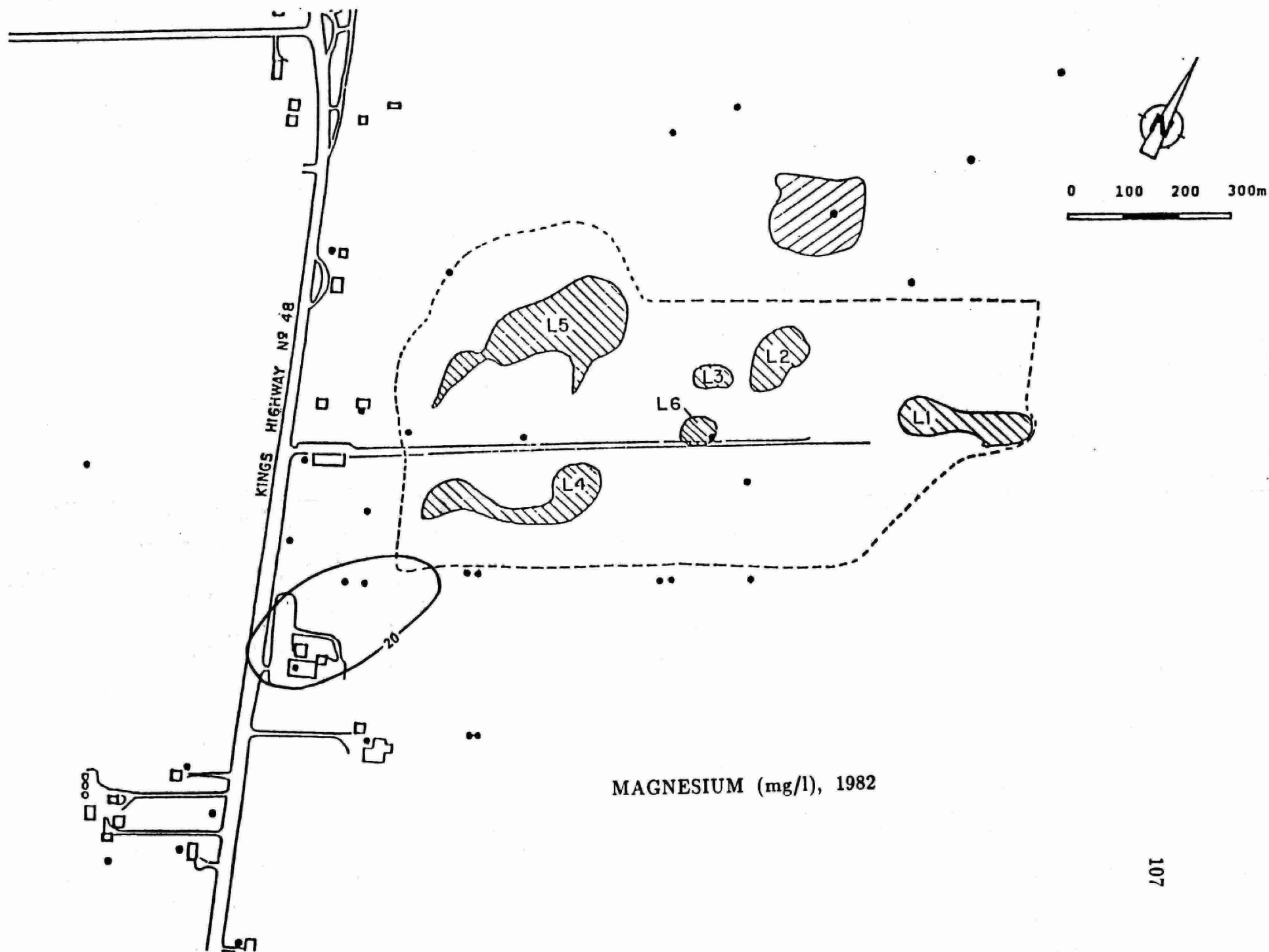
CHLORIDE (mg/l), 1982

KINGS HIGHWAY Nº 48

L5 L2 L3 L6 L1 L7

0 100 200 300m

50 20 50 30 40

SODIUM (mg/l), 1982

SODIUM (mg/l), 1984

KINGS HIGHWAY Nº 48

0   100   200   300m

SODIUM (mg/l), 1986

KINGS HIGHWAY N⁰ 48

0   100   200   300m

L5
L2
L3
L6
L1
L2

50
40
30
30
40
50
60

SULPHATE (mg/l), 1982

104

SULPHATE (mg/l), 1984

KINGS HIGHWAY Nº 48

L5

L3

L2

L6

L4

L1

0   100   200   300m

SULPHATE (mg/l), 1986

KINGS HIGHWAY Nº 48

L5

L2

L3

L1

L6

L4

20

MAGNESIUM (mg/l), 1982

0   100   200   300m

107

KINGS HIGHWAY Nº 48

L5

L2

L3

L6

L1

L2

25

20

MAGNESIUM (mg/l), 1984

0   100   200   300m

MAGNESIUM (mg/l), 1986

POTASSIUM (mg/l), 1982

KINGS HIGHWAY Nº 48

POTASSIUM (mg/l), 1984

0   100   200   300m

L1
L2
L3
L4
L5
L6

POTASSIUM (mg/l), 1986

KINGS HIGHWAY Nº 48

112

CALCIUM (mg/l), 1982

KINGS HIGHWAY Nº 48

L5
L2
L3
L6
L1
L7
120
100
120
100

0 100 200 300m

CALCIUM (mg/l), 1984

CALCIUM (mg/l), 1986
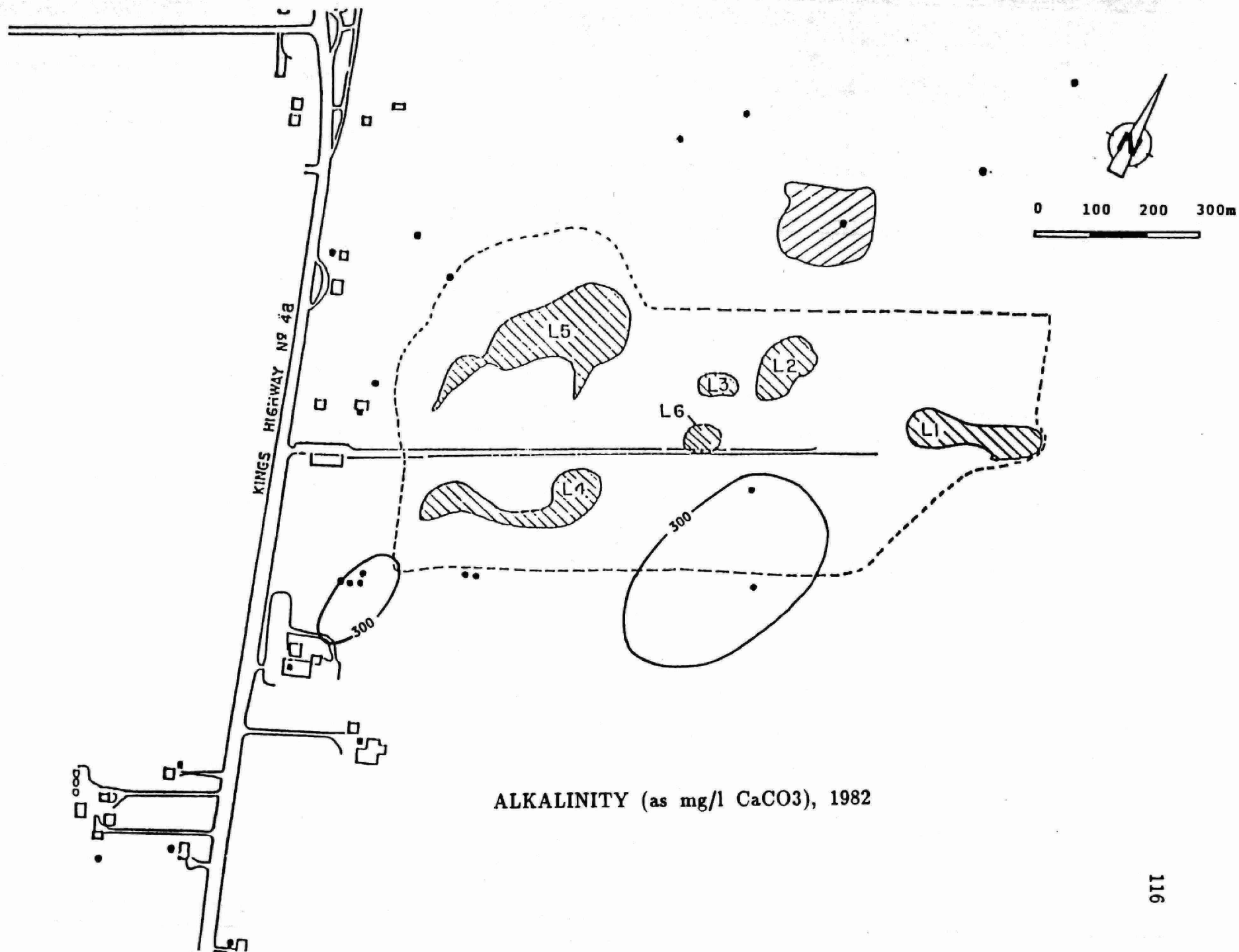
KINGS HIGHWAY Nº 48

0   100   200   300m

KINGS HIGHWAY Nº 48

L5

L2

L3

L6

L4

L1

300

300

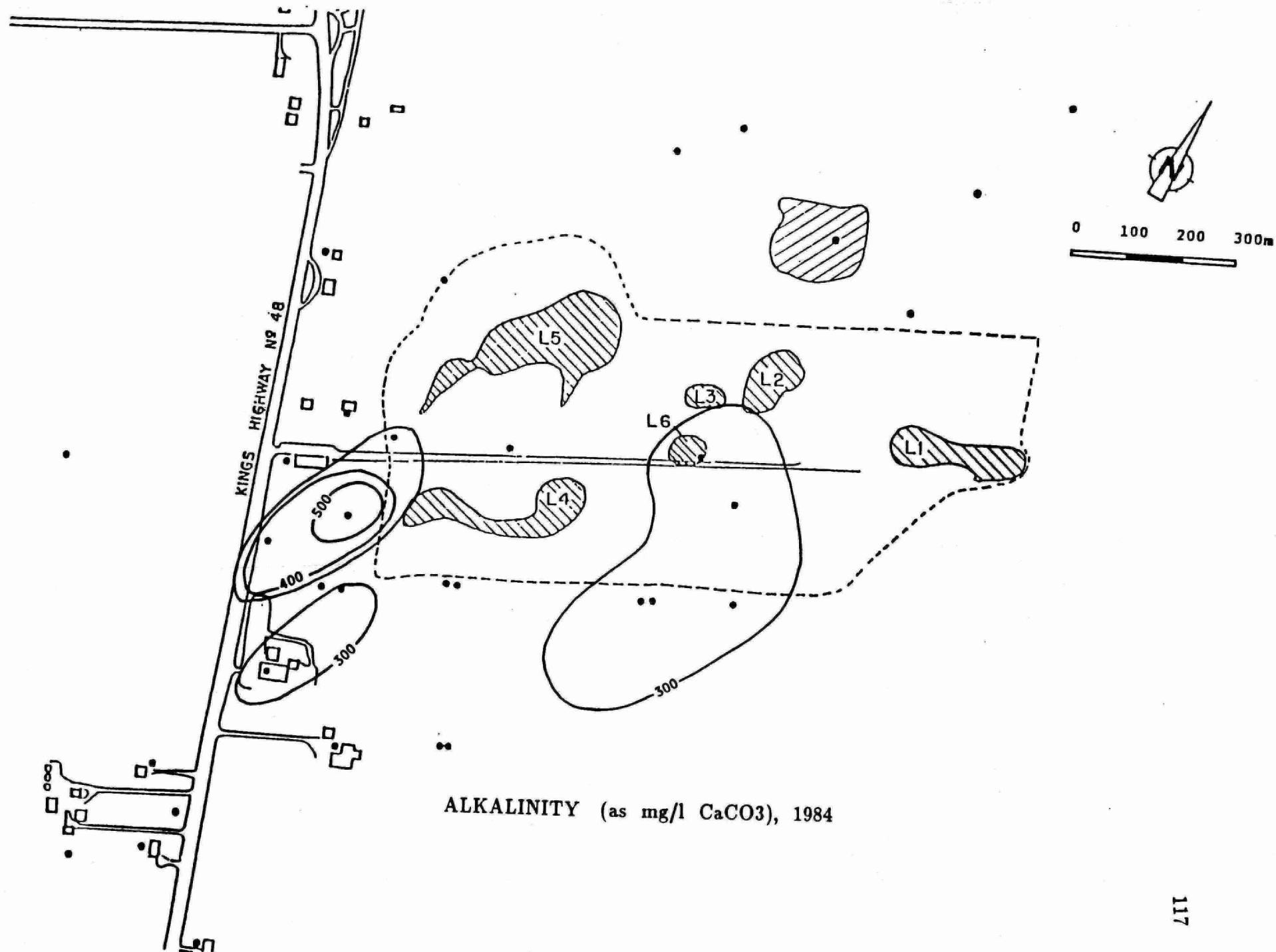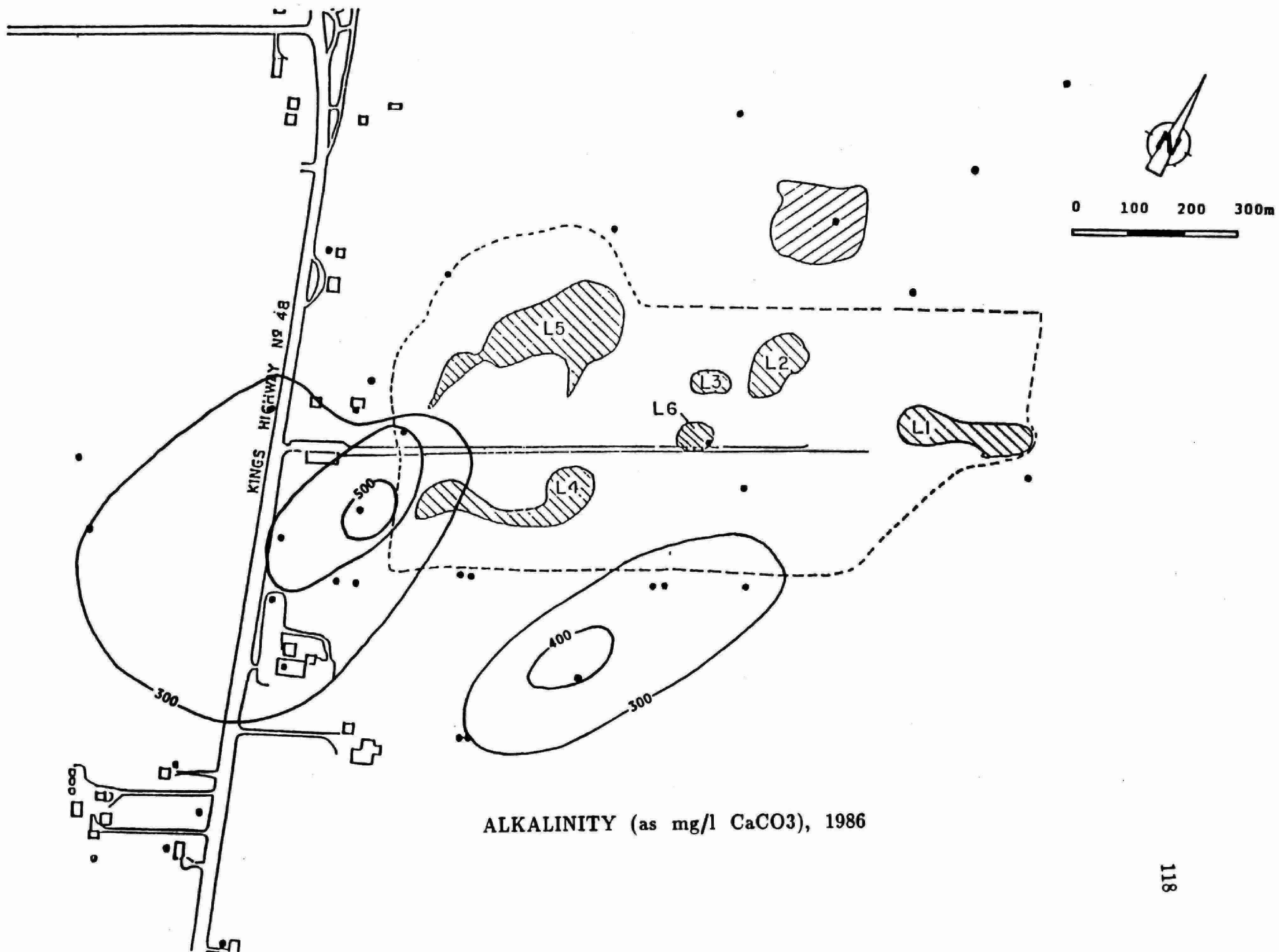ALKALINITY (as mg/l CaCO3), 1982

0   100   200   300m

116

ALKALINITY (as mg/l CaCO3), 1984

KINGS HIGHWAY Nº 48

0   100   200   300m

L5

L2

L3

L6

L4

L1

500

400

300

300

ALKALINITY (as mg/l CaCO3), 1986

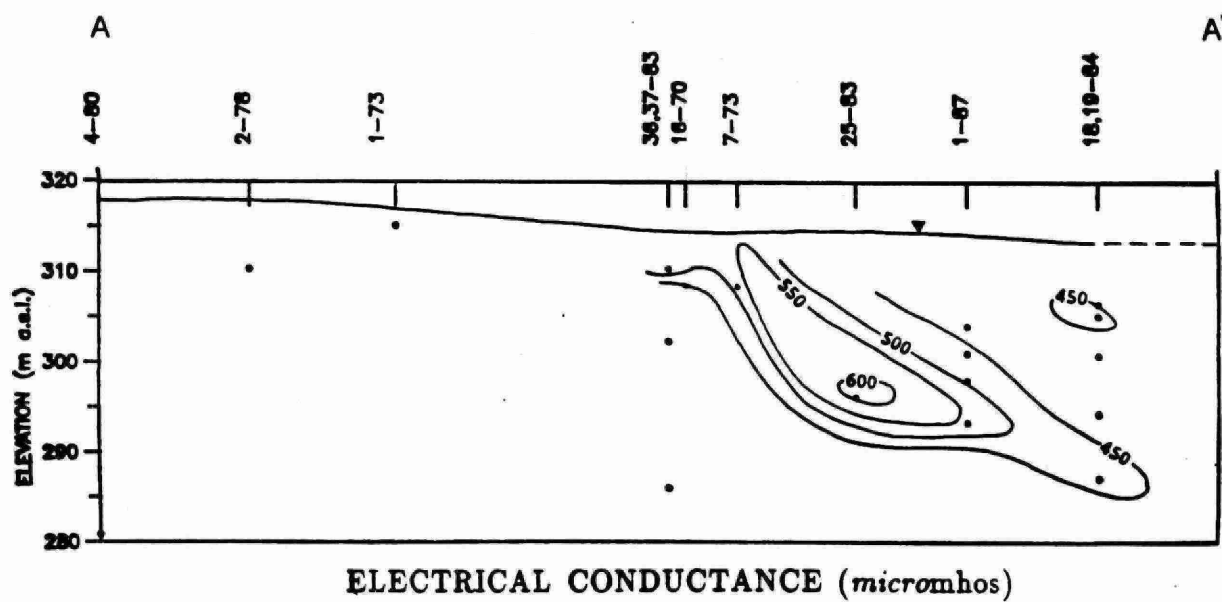DISSOLVED ORGANIC CARBON (mg/l), 1986

Table 5: Values used to delineate the plume in cross section A-A'

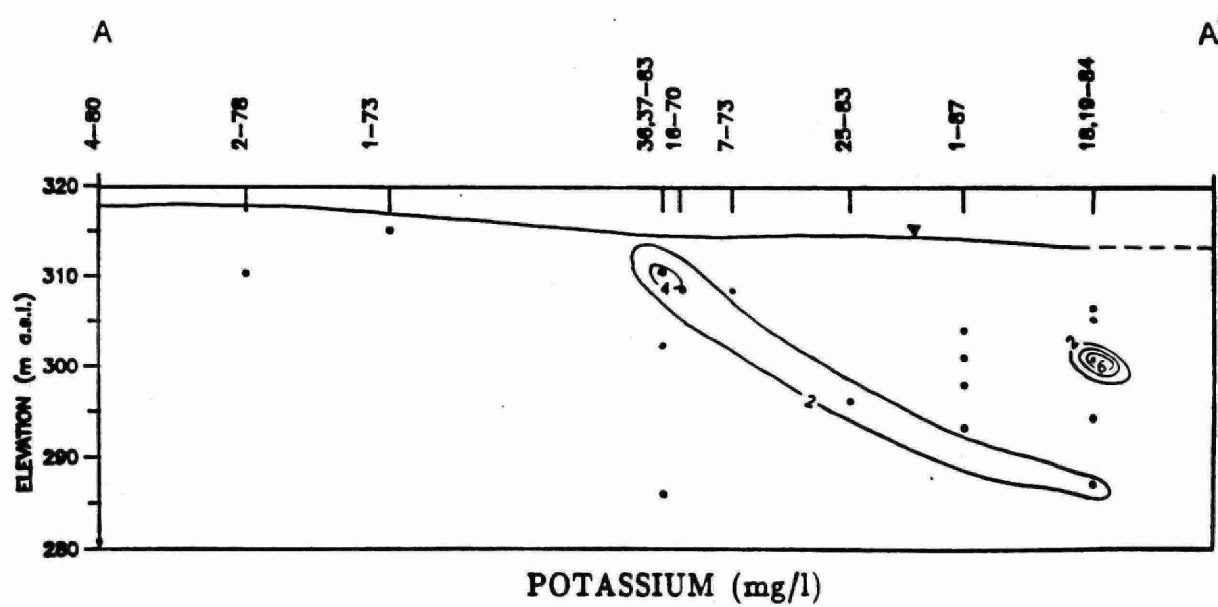| Well	Cond.	Na	SO4	Ca	DOC	Mg	K	Cl
4-80	255	4.6	22.0	57.5	1.4	8.7	1.0	8.1
2-78	350	5.3	18.0	87.2	5.0	16.0	1.6	8.8
1-73	512	19.5	41.0	107.3	8.0	12.8	2.4	33.8
36-83b	255	19.5	3.5	46.7	7.8	12.1	1.9	7.1
37-83a	233	4.2	12.8	57.7	2.4	12.7	1.2	
37-83b	663	12.5	20.0	157.0	7.3	26.4	4.0	12.0
16-70	443	13.4	20.3	99.6	3.1	9.8	2.2	38.9
7-73	533	15.1	20.5	128.0	3.1	20.7	1.2	51.5
25-83b	639	65.3	38.8	97.0	4.6	17.5	2.7	81.8
1-87a		58.3	24.5	16.4	8.5	2.1	0.9	32.0
1-87b		77.8	35.0	27.9	11.0	4.7	1.5	32.3
1-87c		67.4	35.9	18.1	8.5	2.6	0.8	22.9
1-87d		58.0	29.8	14.6	9.0	2.1	0.7	17.2
19-84a	334	5.7	2.7	66.5	3.0	13.4	1.5	17.9
19-84b	458	20.7	2.0	96.4	1.9	10.5	1.7	39.6
18-84a	485	34.0	30.0	76.5	2.5	7.9	2.5	71.4
18-84b	242	27.0	47.0	17.0	4.0	1.3	6.1	16.1
18-84c	440	22.3	5.8	99.0	1.7	14.9	1.3	41.7

Note: all values in mg/l, except conductivity in umhos/cm,
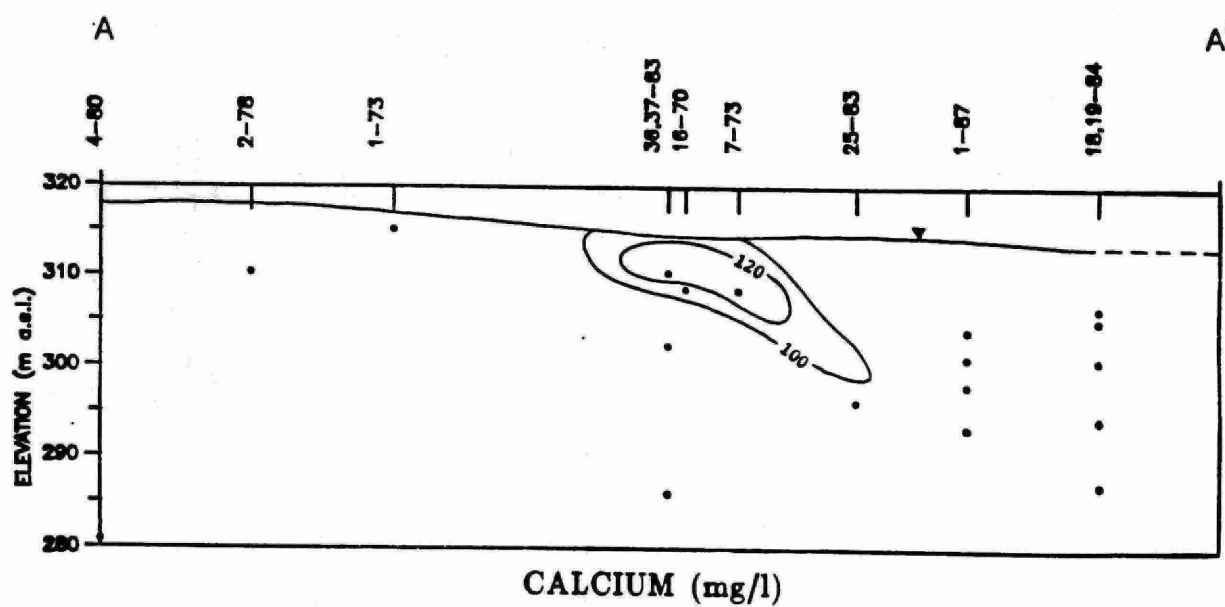average values for 1986 except for 1987 wells

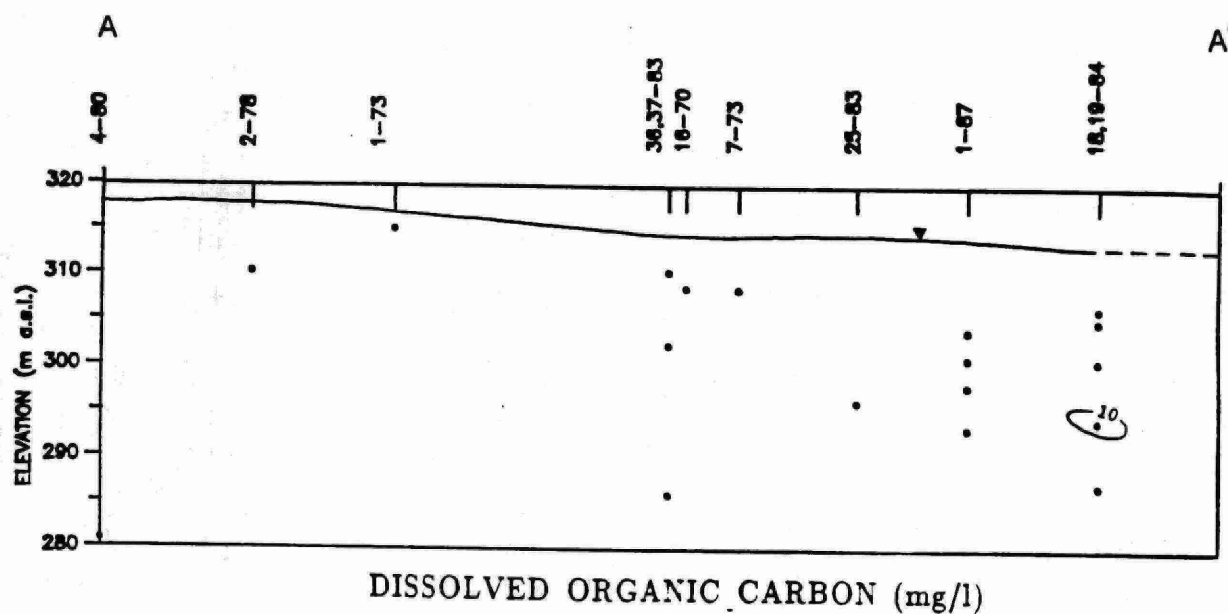CHLORIDE (mg/l)

ELECTRICAL CONDUCTANCE (*micromhos*)

SULPHATE (mg/l)

SODIUM (mg/l)

MAGNESIUM (mg/l)

POTASSIUM (mg/l)

DISSOLVED ORGANIC CARBON (mg/l)

CALCIUM (mg/l)
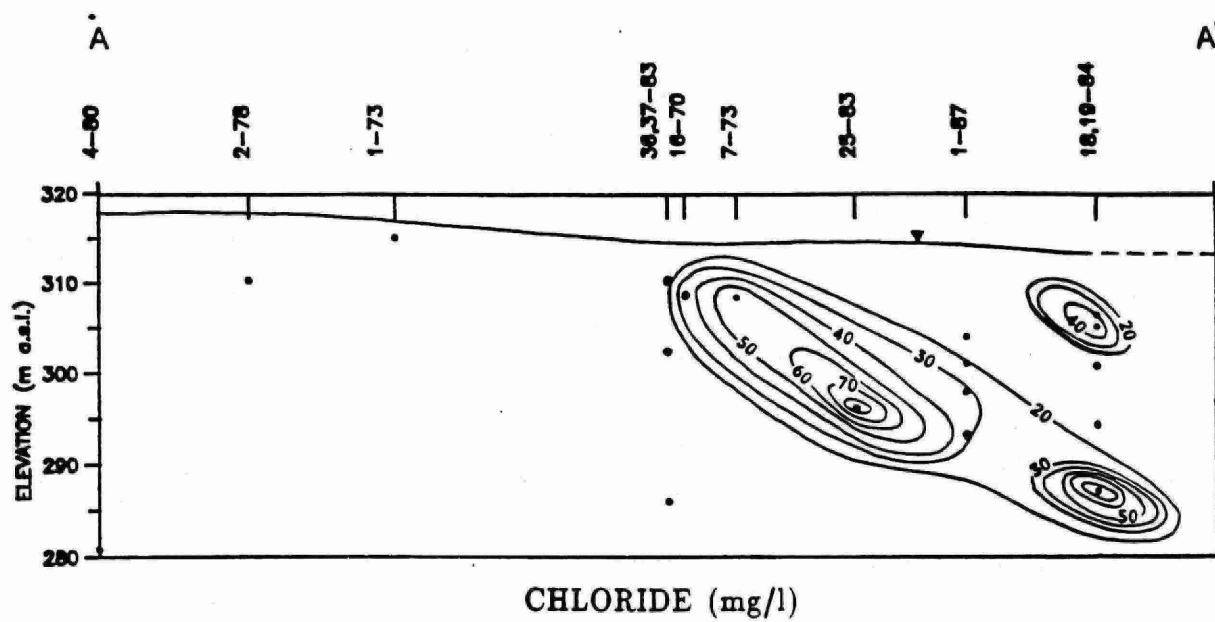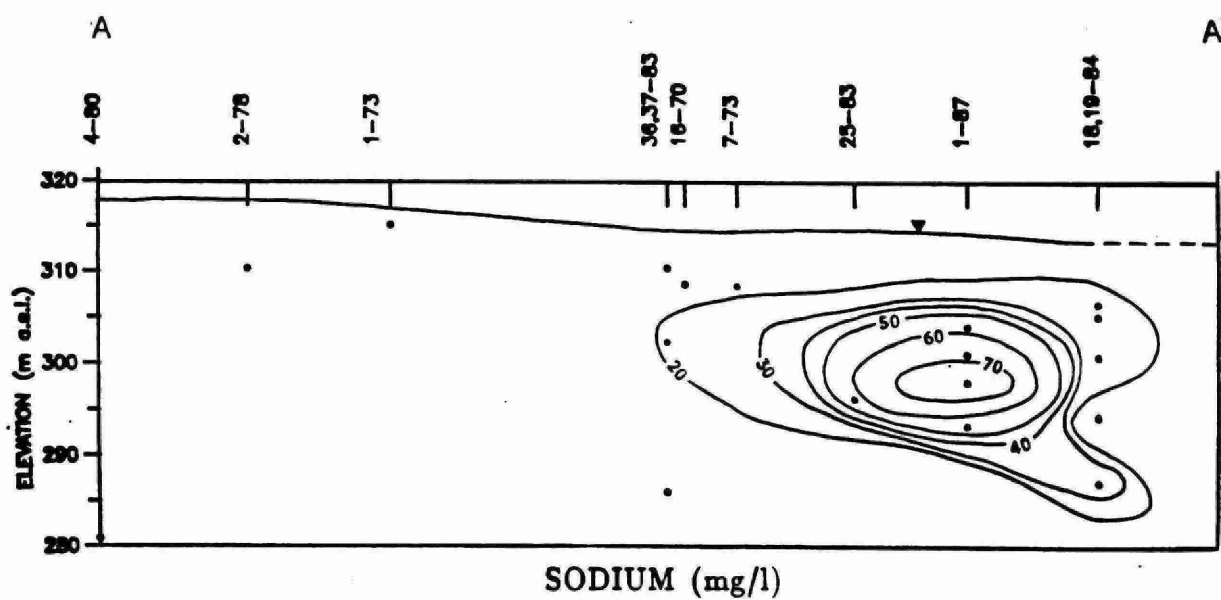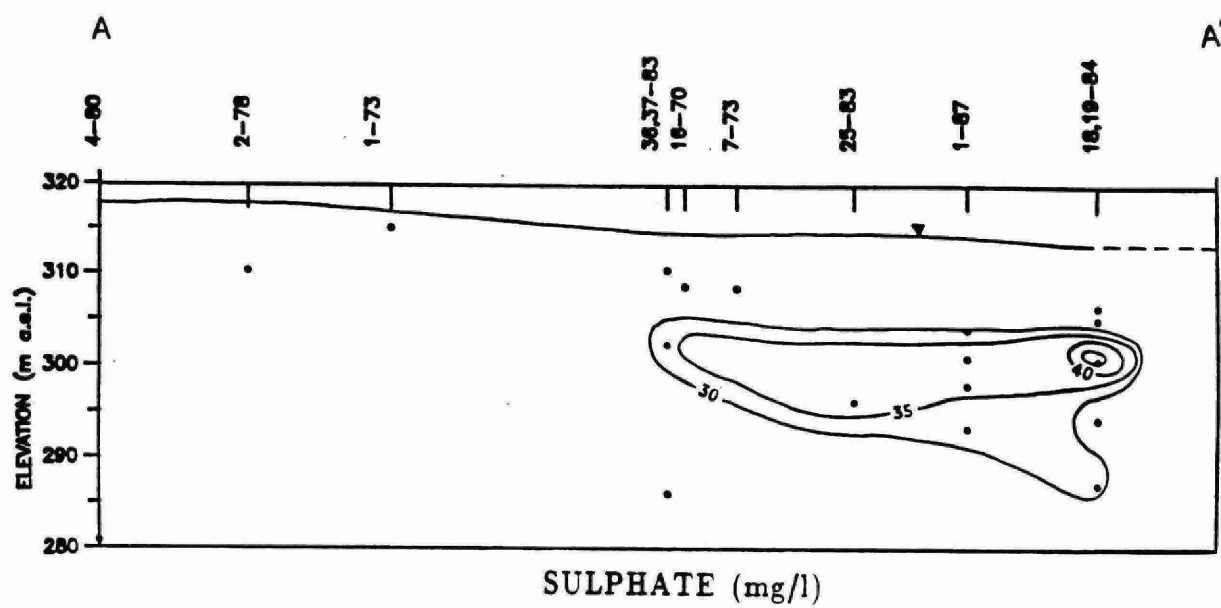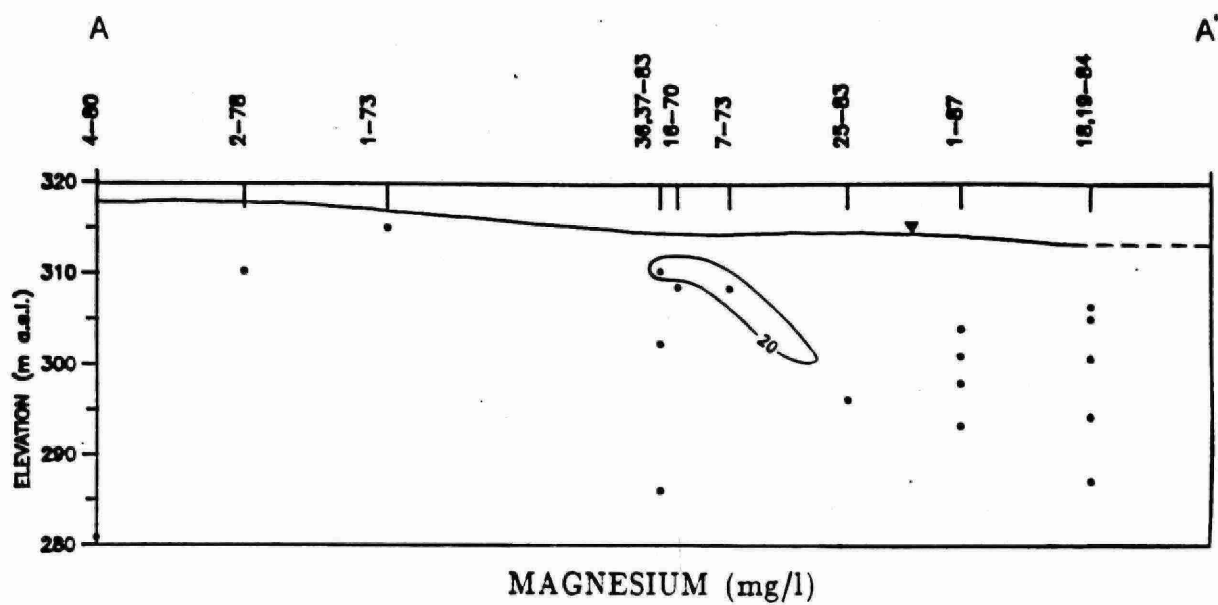
Table 6: Values used to delineate the plume in cross section B-B'

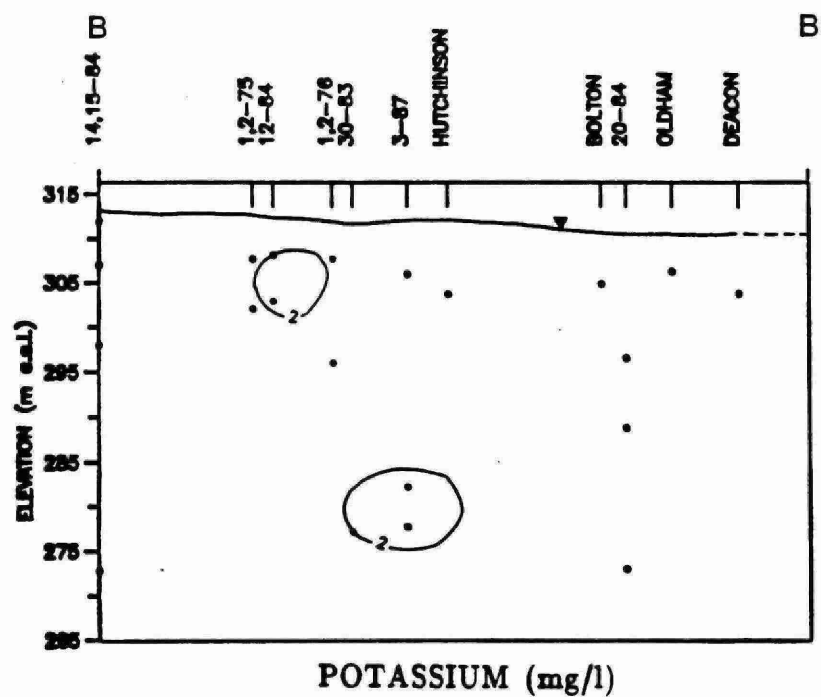| Well	Cond.	Na	SO4	Ca	DOC	Mg	K	Cl
2-75	466	22.5	24.3	69.0	5.0	15.9	1.3	46.6
12-84a	891	97.8	31.0	111.3	18.5	24.3	2.3	66.1
12-84b	769	73.0	21.0	107.8	12.1	25.3	2.4	56.1
1-76	416	14.6	25.5	79.0	2.9	15.3	1.3	32.3
2-76	602	23.3	13.4	104.3	3.0	22.5	1.6	47.4
30-83b	762	138.8	26.8	65.5	18.5	16.7	2.6	82.1
3-87a		39.1	10.1	53.9	11.0	6.1	2.6	50.6
3-87b		29.1	6.6	50.7	10.0	6.7	2.3	30.0
3-87c		48.1	16.6	50.1	20.0	6.7	1.9	42.2
Hutchinson	766	17.1	46.9	129.8	1.1	20.8	1.4	37.6
Bolton	714	37.3	51.7	93.6	1.9	16.1	1.2	50.2
20-84a	535	14.4	84.0	106.0	3.8	12.6	1.6	51.2
20-84b	237	5.8	11.3	50.6	1.8	6.7	1.9	7.0
20-84c	345	5.4	15.4	77.4	1.9	10.1	1.3	16.7
Oldham	461	3.7	19.7	76.9	0.6	12.4	1.1	5.8
Deacon	555	4.2	23.4	96.9	0.7	15.6	1.0	19.6

Note: all values in mg/l, except conductivity in umhos/cm,
average values for 1986 except for 1987 wells
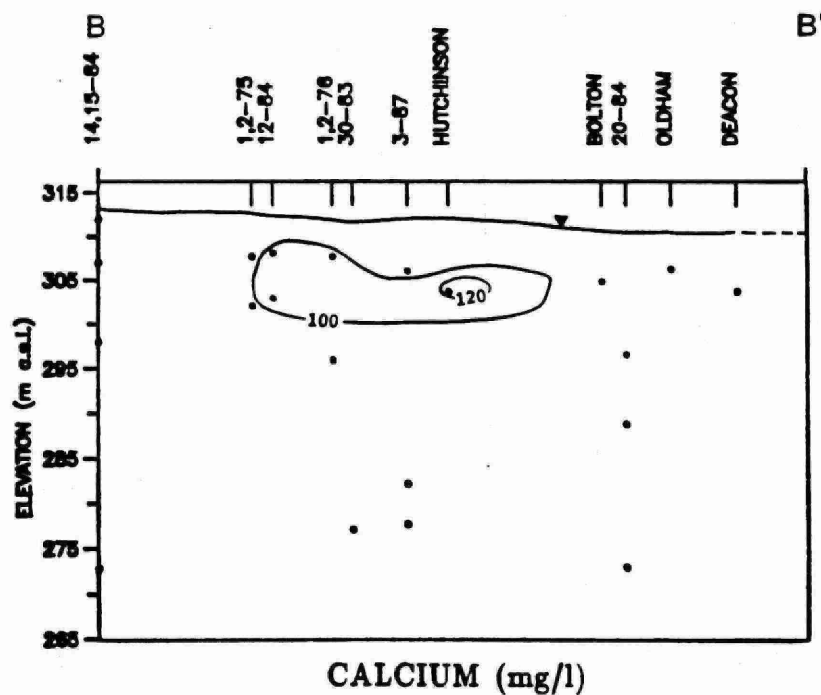
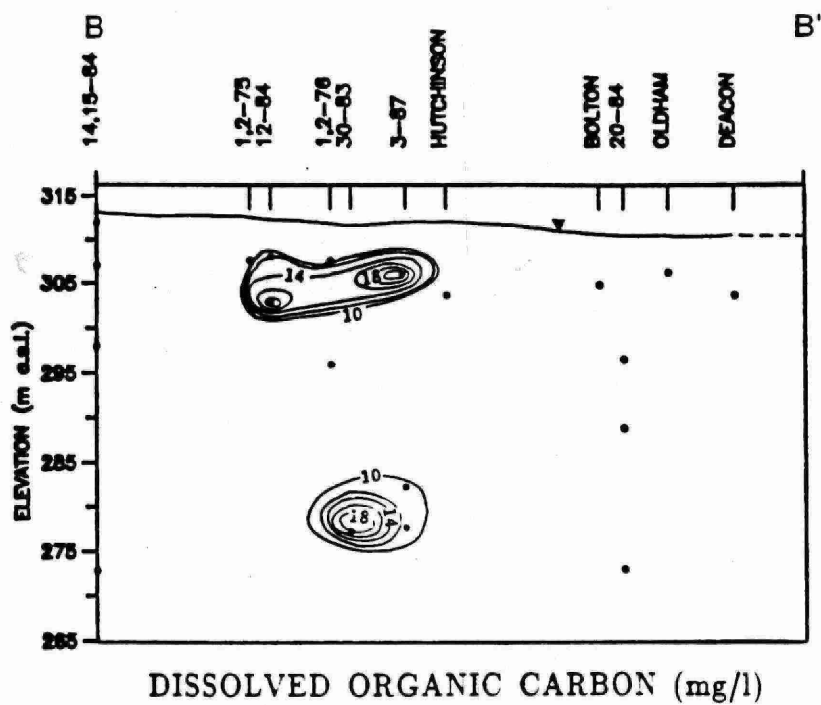CHLORIDE (mg/l)

ELECTRICAL CONDUCTANCE (*micromhos*)

0   100   200   300m

VE = 15.17

SULPHATE (mg/l)

SODIUM (mg/l)

MAGNESIUM (mg/l)

POTASSIUM (mg/l)

VE = 15.17

DISSOLVED ORGANIC CARBON (mg/l)

CALCIUM (mg/l)
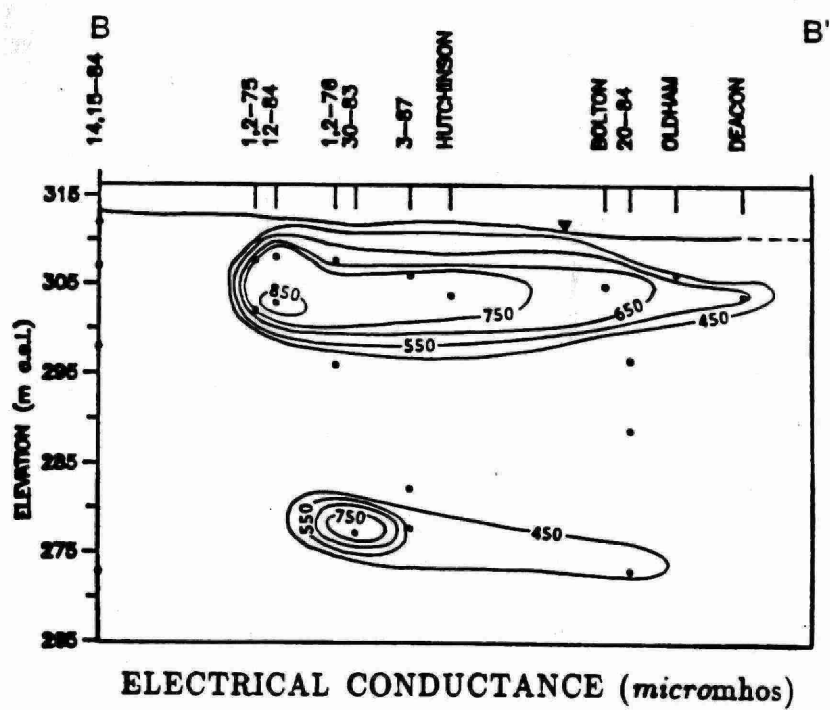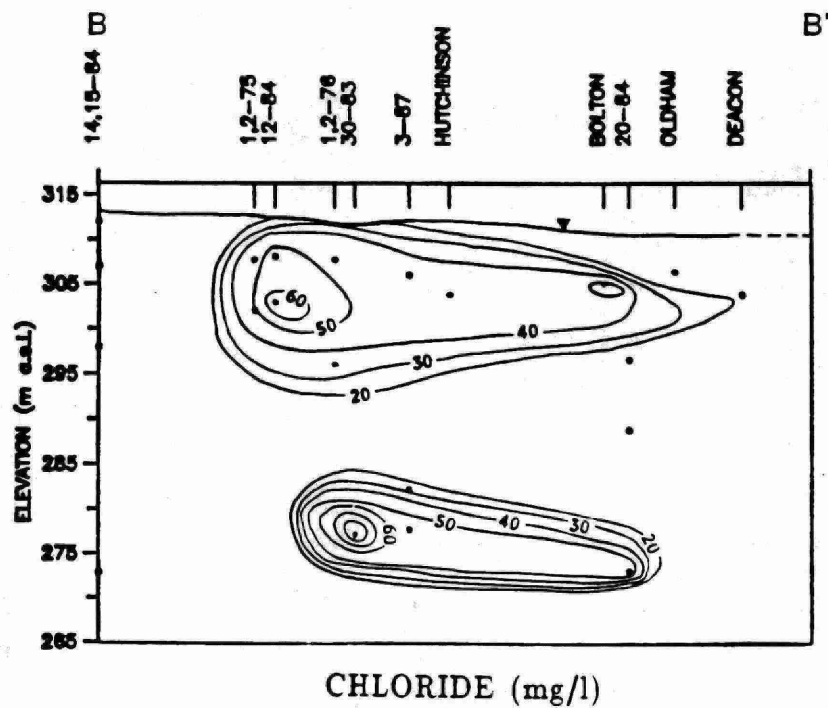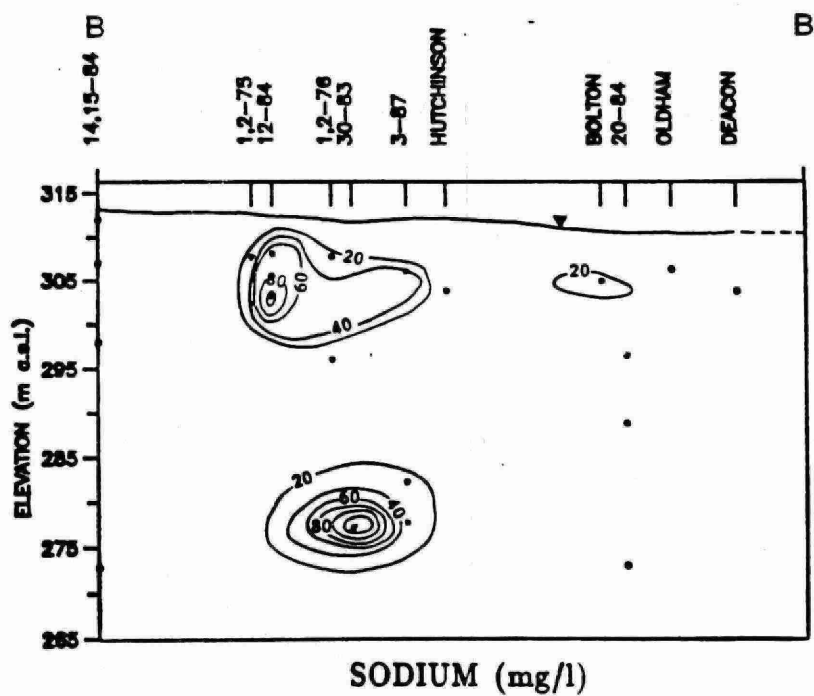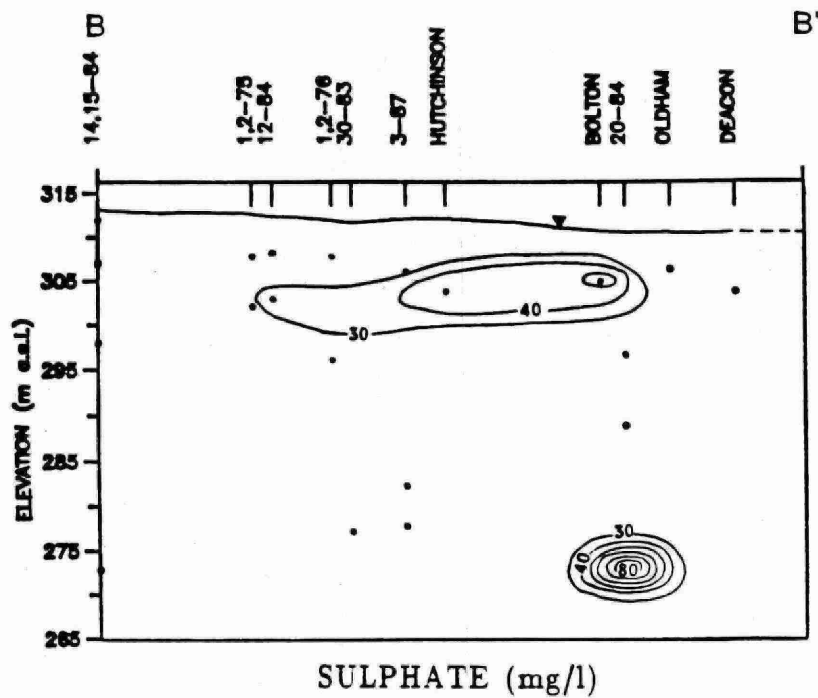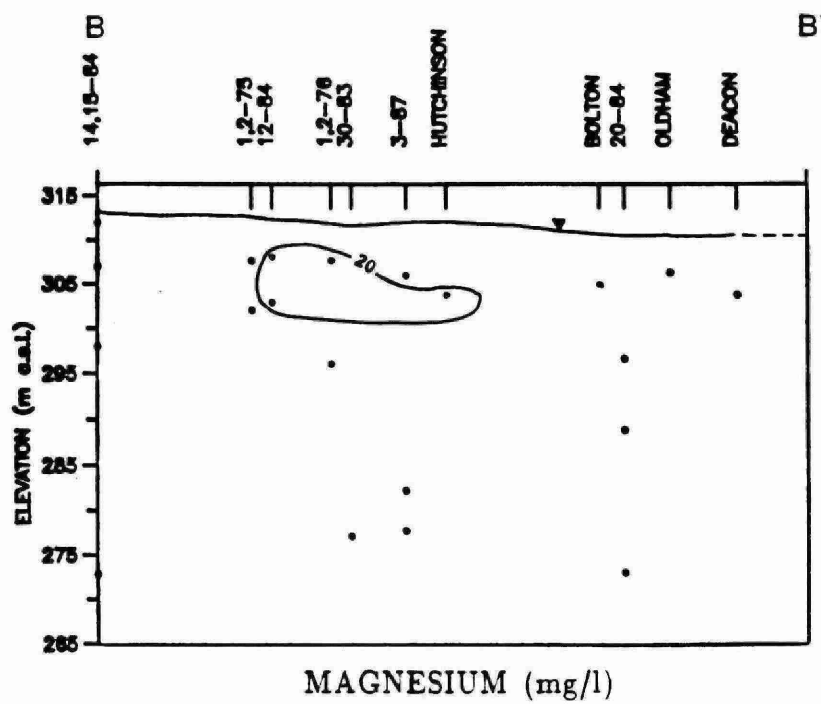
Table 7: Input values for WATSPEC

Well	4-80	37-83b	16-70	7-73	25-83b	1-87b	18-84b
pH	7.62	7.15	7.46	7.48	7.52	7.50	10.59
Ca	1.43	3.92	2.49	3.20	2.43	0.70	0.43
Mg	0.36	1.09	0.40	0.85	0.72	0.19	0.05
Na	0.20	0.56	0.58	0.66	2.84	3.39	1.18
K	.026	.102	.057	.032	.069	.038	.157
Cl	0.23	0.34	1.09	1.45	2.3	0.91	0.46
HCO3	3.34	9.38	5.11	6.99	6.57	4.11	1.12
SO4	0.23	0.21	0.21	0.21	0.40	0.36	0.49

Note: Average concentrations in mmol/l in 1986,
temperature of 10 Celsius

Table 8: Results from speciation program WATSPEC

Well	4-80	37-83b	16-70	7-73	25-83b	1-87b	18-84b
SI calcite	0.0	0.4	0.3	0.6	0.4	-0.3	1.2
SI gypsum	-2.2	-2.0	-2.1	-2.1	-1.9	-2.3	-2.4
SI dolomite	2.4	1.5	2.1	2.1	2.0	2.2	6.6
Log PCO2	-0.7	0.0	-0.4	0.5	0.1	-1.3	1.5

Note: SI is expressed as log [activity ratio], a positive value
indicates supersaturation, 0 equilibrium, and a negative value
undersaturation.

Appendix D

MULTILEVEL PIEZOMETERS - 1987

A bundle of multilevel piezometers consist of 3 or 4 1.27 ID PVC pipes to which are attached 1.5 m long screens made of slutted PVC pipe wrapped with nylon mesh. The piezometers are attached around a 5 cm ID PVC pipe to render the apparitus more rigid and ease this emplacement in the borehole.

The multilevel piezometers were installed in 15 cm diameter holes drilled by a direct mud rotary rig. The bundle of piezometers was put down the hole full of water containing as little mud as possible; the hole was then extensively flushed with clean water. The hole caved around the installation overnight. A bentonite plug was installed under the till unit and the hole backfilled with cuttings. The piezometers were flushed several times using a nitrogen gas line after completion and a week later. Sampling of the piezometers was done after flushing 3 to 4 well volumes with the nitrogen gas line. A bailer consisting of a long polyethylene tube with a one way valve at its end was used to recover the water samples.

Borehole No.   OW 1-87

Location: Diadamo property
Elevation: 1122 feet

Depth (feet)   Description   Installation

0

20   Clay

40

60   Sand   Backfilled
                                              with cuttings

80   Gravel
                                              Bentonite

100

120

            Sand

140

160

            Clay   Backfilled

180

Depth (feet)   Description   Installation

0 —
          Brown sand

          Brown clay
20 —

          Gray clay
40 —

60 —

          Fine sand and silt

          Gray clay
80 —

100 —

                                                  Backfilled
                                                  with cuttings

                                                  Bentonite
120 —

          Brown sand

140 —

160 —

180 —

          Fine sand and silt
                                                  Backfilled
200 —

Depth (feet)	Description	Installation

0 — Brown clay and black peat

20 — Gray clay and silt

40 —

60 — Backfilled with cuttings

80 —

100 — Sand — Bentonite

120 — Gravel

Brown sand

140 —

160 — Gray clay

180 — Silt

200 —

Fine silty sand brown

220 — Backfilled

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